The Ecology of Water Vole (*Arvicola amphibius)* in Grassland Habitats in the City of Glasgow

Robyn Ann Stewart BSc (Hons)

Submitted in fulfilment of the requirements for the degree of MSc of Research

School of Life Sciences

College of Medical, Veterinary and Life Sciences

University of Glasgow

December 2015

I declare that except where explicit reference is made to the contribution of others that this dissertation is the result of my own work and has not been submitted for any other degree at the University of Glasgow or any other institution.

Signed:

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ABSTRACT

The water vole, *Arvicola amphibius*, is a declining British species. Its range is limited to riparian margins along water courses and reed beds and they are considered to have strict habitat preferences. Unusual populations living in grassland habitats in the East End of Glasgow came to light in 2008. This behaviour is largely unrecorded and poorly understood in British populations although grassland populations, termed fossorial, are common in some regions of Europe. The aim of this project was to update current surveying methodology which focuses on riparian habitat, map the distribution of grassland water vole populations in the East End and investigate habitat preference. An area of $34km^2$ was surveyed using stratified sampling methodology and 100m presence/absence transects based on the identification of field signs. A total of 65 sites were identified; 65 were surveyed in March-April and 62 repeat-surveyed in Sept-Oct 2014. Of these 21 were occupied by water voles in March-April and 19 occupied in Sept-Oct. Water vole distribution was concentrated along a 3km stretch of the M8 corridor and adjacent grassland patches. Distribution of occupied sites was linearly related to distance from the M8 corridor with 62% of occupied sites less than 1km distant.

Logistic regression modelling revealed that habitat type and distance from riparian habitat were key indicators in grassland water vole distribution. The distribution of water voles was not related to distance from riparian habitat: sites between 0-150m and sites over 550m distant had equal likelihood of occupation. Only sites at the intermediate distance of 151-550m were less likely to be occupied. Six out of the 9 breeding colonies recorded were over 550m from riparian habitat and at a maximum distance of 1182m.

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Water vole occupation was associated with urban habitats with parkland being the preferred habitat type over road verges and rank grassland. Parkland was characterised by heavy management regimes, moderate to high disturbance and low botanical diversity. The occurrence of water voles was strongly associated with certain grass species, particularly *Holcus lanatus* and *H. mollis* which were the dominant grasses on 43% of all occupied sites and 67% of breeding sites. *Holcus* grasses were also associated with the park habitat type. The main predators of East End populations were fox (*Vulpes vulpes*) and the domestic cat (*Felis catus*). American mink (*Neovison vison*) were never recorded.

Trapping transects allowed for capture-mark-recapture at two sites and the estimation of population size by population modelling in Program MARK. Population densities were estimated at up to156 water voles per hectare indicating grasslands are valuable habitat for water voles.

Their ecological distinctiveness and high densities provide strong evidence that the East End is a key regional stronghold for water voles and that the populations of Glasgow are of national significance. The need for an urgent re-think of current species management guidelines, mitigation protocols and standard surveying methodology has been highlighted by this research.

CONTENTS

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2. INTRODUCTION

The water vole, *Arvicola amphibius*, is a small mammal belonging to the rodent family. Stocky-bodied in appearance, with a blunt muzzle and short tail, it has a broad distribution across Europe, including Britain, which extends into Russia (Corbet & Harris, 1991). Water voles typically inhabit wetland areas, are semi-aquatic and considered a sign of a healthy wetland ecosystem (Stoddart, 1970) although they show considerable variation across their range, with an alternative ecotype found in some regions of Europe which live in grassland habitat (Meylan, 1977). In contrast to the "traditional" semi-aquatic lifestyle of the riparian water vole, populations living in grasslands have a more mole-like existence, termed fossorial, and exist independently of water bodies (Meylan, 1977). The species is classed as Least Concern by the IUCN due to stable numbers across the majority of its range (IUCN, 2014). Populations in the UK are an exception, however, and are protected having undergone a catastrophic decline since the 1950's (Strachan & Moorhouse, 2006).

Original Linnaean classification described both *A. amphibius* and *A. terrestris* (1758) but *A. terrestris* has been the largely favoured species name. These two species are now widely regarded as conspecific (Wilson & Reeder, 2005) and *A. amphibius* has been given naming priority (Strachan *et al*. 2011). Recent work by Panteleyev (2001) split *A. amphibius* into two separate species, *A. amphibius* and *A. scherman*, based on morphometric variation in body size, dentition, pelage and habitat preference which was further supported by a comprehensive review of the species' classification by Wilson and Reeder (2005). *A. amphibius* is larger, with ortodont incisors, coarse pelage and semi-aquatic in nature, whereas, *A. scherman* is smaller, with forward-projecting incisors on the upper jaw and is exclusively fossorial (Wilson & Reeder, 2005). Analysis of mitochondrial DNA polymorphism revealed three distinct lineages within *A. amphibius:* the widespread semi-aquatic group found throughout the species' range, a transitional group of both riparian and fossorial forms south of the Alps and the fossorial form, which is restricted to mountainous areas of the Pyrenees and Alps, (Wust-Saucy, unpublished; Taberlet *et al*. 1998). *A. amphibius* colonised Britain during the last Ice Age (Piertney *et al*. 2005). Genetic work by Piertney *et al*. (2005) found considerable genetic divergence within UK populations and the evolutionary history inferred from parsimony points toward two separate colonisation events with Scottish populations arising from French/Swiss/Spanish colonisers via the Iberian refugium and English populations arising from an Eastern European refugium.

The Fossorial Ecotype

The term ecotype was first proposed by Turesson (1922) to describe the phenomenon of subpopulations arising within a species due to variation of environmental conditions across its range leading to genetic, morphological or phenological adaptations (Turrill, 1946). With its smaller body size, the adult body mass of the fossorial water vole ranges from 60-150g compared to 140-350g for adult riparian water voles in the UK (Saucy, 1994; Strachan *et al*. 2011). It favours upland meadows and can be found at high population densities in mountain regions (Berthier *et al*. 2014). Distribution is not limited by water features and burrow systems can exceed 100m in length in a complex array of runs, nest chambers, food stores and bolt holes and will usually house a single breeding unit, a male and female, with their offspring (Meylan, 1977). Their existence is almost exclusively subterranean, foraging for rhizomes, tubers and fleshy roots along the runs (Meylan, 1977). Plugging up entrance holes with soil is a well-documented behaviour (Meylan, 1977) as is the creation of above ground soil mounds, termed tumuli, a by-product of digging activity (Giraudoux *et al*. 1995).

Fossorial populations can become a serious pest in some areas due to the economic impact they can have on agricultural crops and orchards by damaging root systems, consuming plants and digging extensive burrow systems which can destabilize soil structure (Meylan, 1977). Populations oscillate with a 4-8 year cyclicity and in peak years water vole numbers can reach "outbreak" proportions of a 1000 individuals per hectare (Giraudoux *et al*. 1995; Weber *et al*. 2002; Berthier *et al*. 2014). Giraudoux *et al*. (1995) noted that peak year populations were forced to expand into surrounding habitats even if they were unfavourable and regulation of the population is thought to be largely down to density-dependent factors such as food availability and disease (Saucy, 1994).

The Riparian Ecotype

It is thought only the riparian ecotype occurs in Britain (Corbet & Harris, 1991) with a rare handful of accounts of fossorial behaviour (e.g. Read's Island, Eilean Gamhna) although recent studies by Telfer *et al*. (2003a) identified large populations of fossorial water voles on a number of small islands in the Sound of Jura.

Riparian water voles have highly specialized habitat requirements and their distribution is limited by the availability of riparian habitat, which normally exists as a narrow band of wetland vegetation following the length of a water course, most commonly rivers and streams, in a linear pattern (Stoddart, 1970; Lawton & Woodroffe, 1991; Strachan *et al*. 2011). Non-linear wetland habitats like reed beds have also been found to support extensive water vole populations (Carter & Bright, 2003).

Water voles are herbivores with a broad diet, feeding on over 227 species of vegetation (Strachan *&* Moorhouse, 2006). They are believed to favour tall grasses, rushes and sedges e.g. *Phragmites australis*, *Carex* sp., *Deschampsia* sp. and *Molinia caerulea* (WildCRU, 2002) and leave characteristic piles of foraging remains at feeding stations throughout their home range (Strachan & Moorhouse, 2006).

Water vole populations consist of multiple breeding units strung out along the length of the water course with females being the territorial sex during the breeding season, demarcating the area with piles of droppings called latrines and actively excluding other females, in contrast to the larger home range of the males (Strachan & Moorhouse, 2006). The length of habitat occupied is dependent on population density with mean territory size measuring 30m to 150m for females and 60m to 300m for male home ranges at high and low densities respectively (Strachan & Moorhouse, 2006). The mating season is triggered by increasing day length in early Spring and extends from March through to September (Strachan & Moorhouse, 2006) although breeding as early as February has been documented (Stoddart, 1970). On average females give birth to five to eight offspring and have multiple litters throughout the breeding season. Life expectancy can reach three years but a lifespan of twelve months is far more common and water vole populations are subject to high over-winter mortality rates of up to 70% (Strachan & Moorhouse, 2006).

For some time there has been confusion over the exact habitat preference of water voles, particularly because suitable sites will often go unoccupied (Lawton &

Woodroffe, 1991). It is widely accepted however, that established colonies require a length of continuous riparian habitat, slow-flowing water, soft banks for burrowing, and dense vegetation for both cover and food (Lawton & Woodroffe, 1991; Aars *et al*. 2001; Telfer *et al*. 2001; Lambin *et al*. 2004; Fischer *et al*. 2009). Habitats subject to heavy grazing, trampling or over-shading by tress are actively avoided (Strachan & Moorhouse, 2006). The length of riparian habitat required varies between lowland and upland populations, largely because of habitat quality, with lowland colonies occupying lengths of 100-400m (Lawton & Woodroffe, 1991) and upland colonies occupying lengths between 50-700m although length varies between years (Capreolus Wildlife Consultancy, 2005).

Habitat suitability is intrinsically linked to habitat quality. Habitat preference can be considered an adaptive choice because of the direct relationship often displayed between habitat quality and number of offspring produced (Alcock, 2005). As a result, habitats can be graded in accordance with how well they provide resources; breeding animals require greater resources because of the increase in energetic costs of territoriality and rearing offspring (Lawton & Woodroffe, 1991). Habitat quality also influences territory size with better quality habitats supporting a greater abundance of water voles (Strachan & Moorhouse, 2006).

Conservation status

Water voles are considered to be one of Britain's fastest declining mammal species and national surveys conducted by the Vincent Wildlife Trust show that since 1990 there has been an overall UK decline of 88% (Strachan, 2004). As such, the water vole is classed as a UK Biodiversity Action Plan priority species and receives legal protection under Schedule Five of the Wildlife and Countryside Act 1981 and the

Nature Conservation (Scotland) Act 2004 which makes it an offence to intentionally or recklessly disturb, damage or destroy water vole burrows or any structure they may use for shelter (Strachan, 2004). Legislation differs between countries within Britain with the animals themselves given direct protection in England and Wales but not in Scotland (Strachan & Moorhouse, 2006).

Historically water voles have been declining since the early 1900's due to changes in land-use and habitat fragmentation with the move towards intensive agriculture and urbanisation resulting in the loss and degradation of vast areas of riparian habitat (Rushton *et al.* 2000; Strachan & Moorhouse, 2006). Rushton *et al*. (2000) estimate that a third of all semi-natural vegetation once available to water voles on farmland has been lost in the UK since 1940.

The accidental introduction of the American mink (*Neovison vison*) from fur farms in the 1950's resulted in a rapid increase in the rate of loss of water vole populations (Woodroffe *et al*. 1990; Barreto *et al*. 1998; Strachan *et al*. 1998; Carter & Bright, 2003). American mink readily became established because native mustelids like the otter (*Lutra lutra*) and the European polecat (*Mustela putorius)* were largely absent across their normal range and mink are now widespread throughout the UK with the exception of northern Scotland (Harrington *et al*. 2009). American mink are semi-aquatic and hunt along the lengths of water courses and radio-tracking studies have shown they display a high fidelity to water with their daily foraging range being within 10m of a water course (Macpherson & Bright, 2010). They are powerful swimmers and their small body size (males 1.2kg, females 0.7kg) means they can easily fit down water vole burrows, particularly the smaller females, which renders the normal water vole predator evasion behaviour of diving into the water or hiding in their burrows ineffectual (Carter & Bright, 2003). Mink are highly efficient predators of water voles and can be responsible for over 47% of predation on a water vole colony (Carter & Bright, 2003).

Water vole populations already fragmented by habitat loss and degradation became more vulnerable to localised extinction through heavy predation by American mink (Carter & Bright, 2003). Barreto *et al*. (1998) found that the presence of mink was the most important variable in determining water vole distribution in lowland English populations. Modelling of water vole populations by Rushton *et al*. (2000) indicate that small colonies are especially vulnerable and fragmented populations were highly likely to become extinct if mink remove 30% or more of the water vole colony annually. Studies of the River Bynack catchment area, a tributary to the River Dee (Aberdeenshire), showed that the arrival of mink led to a crash in the water vole population within less than a year although small upland populations do experience localised extinction due to environmental and demographic stochasticity (Aars *et al*. 2001).

The rate of decline has shown signs of starting to slow in more recent years, largely in part due to robust mink control programmes but also because of collective efforts between government agencies, statutory nature conservation bodies and wildlife charities resulting in habitat improvement measures and re-introduction projects (Strachan *et al.* 2011). The release of 60 captive bred animals to the River Bude in Corwall in 2013 by the Environment Agency and Westland Countryside Stewards marked a milestone in water vole conservation as for the first time in nearly 30 years, all of England’s counties were occupied by breeding colonies. The Environment Agency also aims to create 10,000ha of wetland habitat by 2020 (Environment

Agency, 2014). Similar stories of tentative recovery are found across Scotland e.g. Insh Marsh (RSPB, 2014) with water voles returning to areas which were occupied prior to the mink invasion after the successful implementation of the Scottish Mink Initiative in 2011 which controls American mink over 28,500 km^2 of the country (Scottish Mink Initiative, 2013).

Scottish Populations

Approximately 40% of the UK water vole population is thought to reside in Scotland with the majority of water vole colonies found as upland metapopulations spread across vast swaths of the Grampian mountain range and Assynt (Stewart *et al*. 1998; Capreolus Wildlife Consultancy, 2005). A metapopulation consists of a series of discrete, small populations linked by the dispersal of individuals which arise because of the patchy distribution of suitable habitat fragments (Hanski, 1991). Upland water vole populations conform to the metapopulation paradigm with colonies being maintained by dynamic extinction and re-colonisation events because of their ability to disperse up to 2km to neighbouring colonies (median distance to nearest neighbouring colony 0.6-0.7km) (Aars *et al*. 2001). Aars *et al*. (2001) two year study revealed that over 30 colonies were connected across an area of 25km^2.

Water vole numbers in the lowlands are considered marginal in comparison, being small and spatially scattered, primarily because of urbanisation and well-established mink populations (Telfer *et al*. 2001). However, Telfer *et al*. (2001) study revealed that lowland populations in the north-east of Scotland also function as a metapopulation with colonies connected by interpopulation dispersal distances of less than 1.3km. Colonies were small in number (median colony size of four individuals and maximum 12) and subject to high levels of extinction. Water voles were also found to disperse directly overland and not be restricted to the water course (Telfer *at al*. 2001).

The Scottish Mink Initiative does not extend south far beyond the Highland boundary fault and American mink are still considered widespread in the lowlands. Urbanisation is largely concentrated in the lowlands with the Central Belt, an area within the lowlands, being the most densely populated containing 80% of the human population in just 6% of the country's total land area (Office of National Statistics, 2011). Such a high human population density means that the majority of the landscape has been modified for housing, recreational areas, infrastructure, agriculture and industrial use. Urbanisation is linked with the loss of vegetation and a reduction in species richness but cities are not always the hostile environment we think them to be with some species adapting and thriving (Pickett *et al*. 2001; McKinney, 2002; Lopucki *et al*. 2013).

Project Background

Cities may not normally be associated with riparian habitat but well managed urban areas can include artificially created parks with water courses/reed bed, remaining sections of canal and wildlife-friendly Sustainable Urban Drainage Systems (SUDS) which can all prove favourable to wildlife and to water voles (Strachan *et al*. 2011). While suitable habitat is patchy, the Forth and Clyde canal, which extends across the Central Belt linking the cities of Glasgow and Edinburgh, does provide stretches of riparian margin for water voles. However, the last official water vole survey commissioned by SNH in 2001 reported only 6 colonies along the 56km surveyed which highlighted a 75% decline in seven years since the previous survey (WildCRU, 2002). The largest number of field signs were recorded at Possil and Firhill, areas to the north of Glasgow. Current estimates of water vole numbers within the city of Glasgow are unknown but riparian populations are considered to be in decline (Cath Scott, *pers comm*.). Given the species protected status and decline in numbers water voles are a Local Biodiversity Action Plan (LBAP) priority species for Glasgow City Council (GCC) and a number of Local Nature Reserves (LNRs) are managed for the benefit of water voles e.g. Hogganfield Park and Robroyston Park by GCC and Cardowan Moss by the Forestry Commission Scotland (FCS).

In 2008 the Natural Environment Officers of GCC were made aware of water vole populations in the city's East End by the accidental capture by the Environmental Health Department of a water vole in traps intended for dealing with a rat infestation. What was highly unusual was that the location – a garden in a housing estate in the Garthamlock area with no known nearby water course or suitable riparian habitat. Further investigation revealed sites with what appeared to be abundant water vole field signs in derelict areas, road verges and public parks. Regeneration of the East End is a high priority for the Scottish Government and GCC because of the higher than national average social deprivation levels in the area (Scottish Government, 2012). To overcome financial constraints GCC have leased council-owned land, including many vacant and derelict sites, to housing developers under the agreement that a proportion of the new properties built will be social housing. However, the East End water voles have taken up residence on numerous sites ear-marked for development which has led to the urgent need to investigate the ecology of these unusual populations. Standard mitigation guidelines focus on displacement techniques of maintaining unfavourable conditions (e.g. preventing water vole expansion into a new area by keeping grass mown short) and translocation in more pressing instances (Strachan, 2004). These guidelines are all based on the riparian water vole which may render aspects of this mitigation inappropriate for the East End populations inhabiting grasslands.

The East End is known to support riparian water vole populations in wetlands areas such as Hogganfield Park and Seven Lochs Wetland Park (GNHS, unpublished) but exactly how some populations have come to move into grassland habitats is largely speculative at this stage. It is possible that the East End water voles are a relic of populations once found along the Monkland Canal and have persisted in the area since the Canal was filled in during the 1950's to create space for the M8 (Scottish Canals, 2015). Reasons why the water voles are being found in grassland habitat could be as follows:

1. Grasslands are unfavourable and only used opportunistically during times of high population density by riparian water voles or by juveniles forced to use grassland to disperse.
2. East End populations are expressing historic fossorial behaviour which originates from their founding European relatives and are capable of utilising both riparian and grassland habitats.
3. The East End populations have moved into grassland habitat as an anti-predator adaption in response to the presence of American mink.

Aims

The aim of this study is to investigate the distribution and ecology of grassland water vole populations in order to address the current lack of knowledge. This information is essential for the local authority, Scottish Natural Heritage, local developers and ecologists and will go toward informing best management practice, surveying techniques, protected species licensing and mitigation guidelines.

Objectives

1. To establish the distribution of water voles across the East End of Glasgow using presence/absence surveys and investigate if distance from water is a key factor in the distribution of grassland populations.
2. To establish key habitat requirements of grassland water vole populations, investigate dominant grass species, disturbance levels and their main predators.
3. To examine the relationship between field sign indices and water vole abundance in grasslands using capture-mark-recapture (CMR) techniques.
4. To identify core breeding sites and establish grassland population density using CMR.
5. To identify the main conflicts affecting grassland water vole populations in the East End of Glasgow.

3 – MATERIALS AND METHODS

3.1 - Distribution of water voles across the East end of Glasgow

3.1.1- Study Area

The Seven Lochs Area (see Fig 3.1) was selected as the study area for establishing the distribution of the water vole across the east of Glasgow. This area is situated 3km east from Glasgow city centre and is demarcated by three trunk roads, the M8, M80 and M73. It is predominately an urban habitat, which contains within it a mosaic of habitat types.

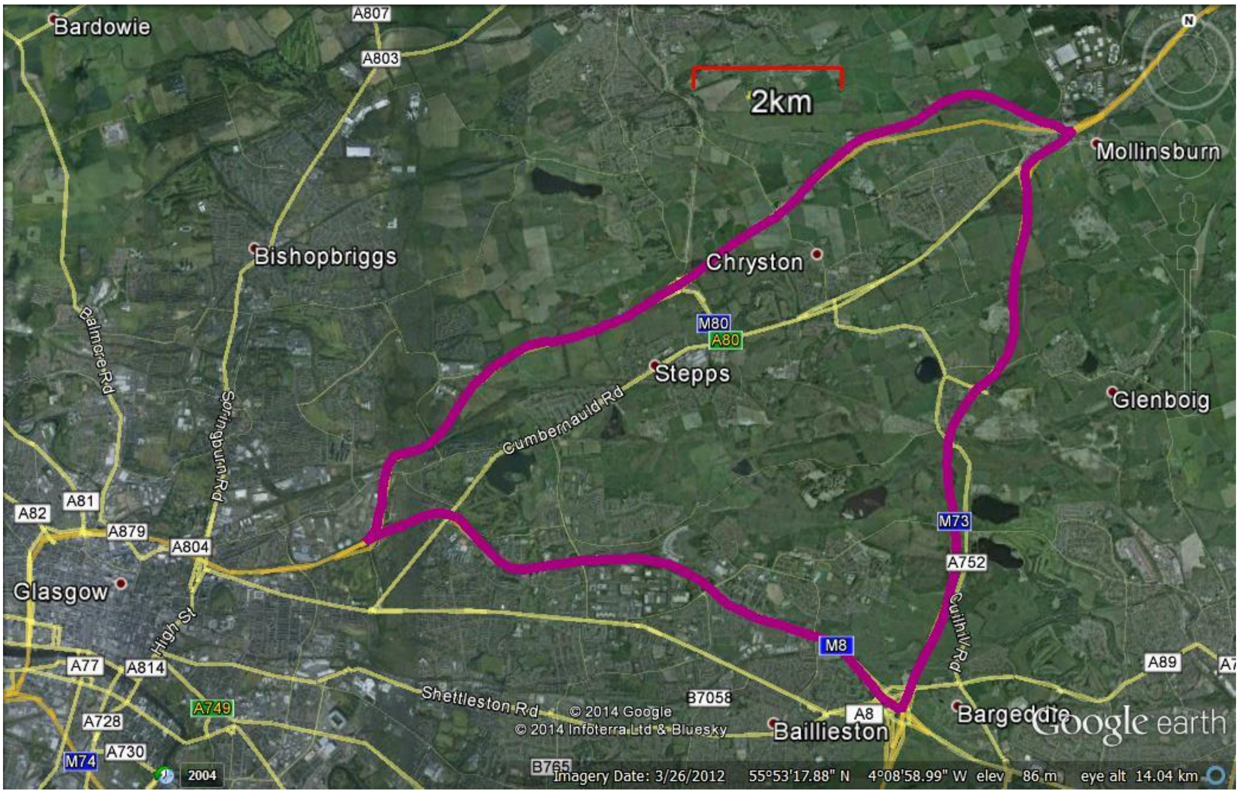

Fig 3.1: Google Earth satellite image of eastern Glasgow with study boundary (purple line).

In total the area encompasses 30 km² of which approximately 17 km² is urban habitat and takes in the city suburbs of Ruchazie, Garthamlock and Easterhouse at the most southern boundary along the M8 and the towns of Stepps, Gartcosh, Chryston and Moodiesburn to the north and east. The Seven Lochs Wetland Park, an actively managed area of man-made lochs, parkland, Local Nature Reserves (LNRs) and woodland (Seven Lochs, 2014) makes up 9 km² within the study site. The Wetland Park extends beyond the M73 further east to Coatbridge but was excluded from the study area due to time budget considerations. North of the Wetlands Park lies predominately agricultural land continuing on to Chryston and Moodiesburn and the M80 boundary.

3.1.2 - Presence/Absence Field Sign Surveys

Primary Sampling Units (PSUs)

A stratified sampling methodology based on The Mammal Society Research Report No. 6 (Sibbald *et al.* 2006) was adopted for surveying water vole presence or absence. Presence was recorded by the detection of field signs (see Section 3.2). The total study area was sub-divided into 30 Primary Sampling Units using grid squares of the OS 1:25 000 Explorer Map 342 of Glasgow. Only 1km² grid squares with over 75% of their total area situated within the Seven Lochs boundary were included in survey efforts. An additional 4 1km² grid squares were included which were outwit the study area boundary based on advice from GCC and initial field visits which showed water vole presence: Robroyston Park, Cranhill Park, and Queenslie Industrial Estate (see Fig 3.2).

Secondary Sampling Units (SSUs)

Each PSU was viewed using Google Earth satellite imagery in order to identify areas of continuous grassland habitat suitable for conducting 100m field sign survey transects, referred to as the Secondary Sampling Unit (SSU). However, given the rate of urban regeneration in the East End of Glasgow, Google Earth proved unsuitable for site selection as the images were not sufficiently up to date. Instead each PSU was systematically walked on foot to locate suitable sites and two SSUs were identified for each 1km² grid square, giving a total of 68 sites. PSU No. 2 had to be excluded due to the lack of suitable habitat.

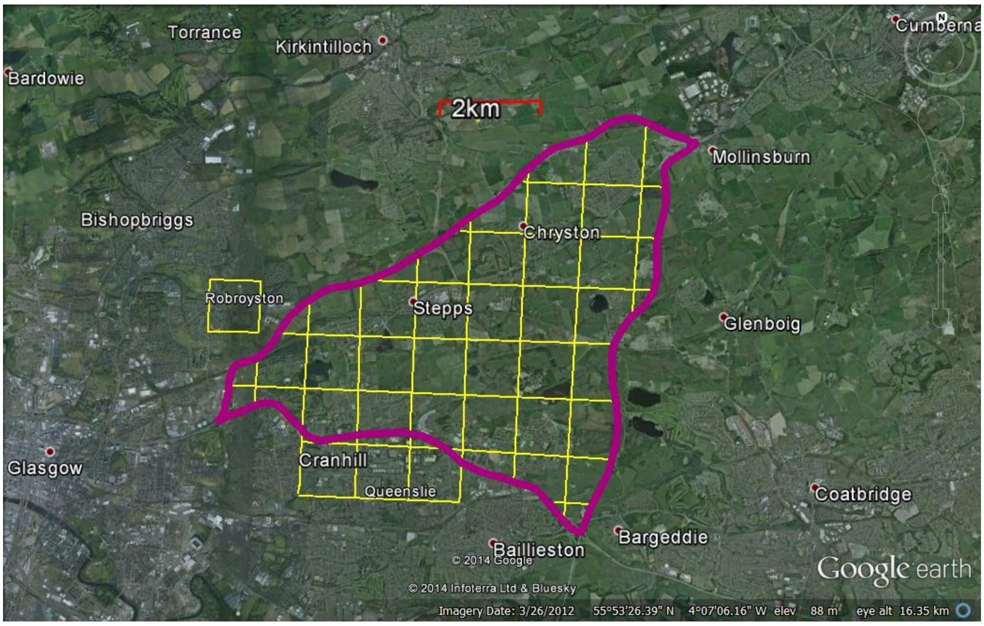

Fig 3.2: Google Earth satellite image of study area and PSUs.

Methodology proposed by Telfer *et al.* (2003a) was adopted for conducting field sign survey transects, with signs within a 2.5 m radius being counted and recorded at each 10m interval along the 100m transect. Field sign transects were non-intrusive and caused negligible disturbance to burrows or other above ground places a water vole may occupy, therefore, fully complied with current wildlife legislation.

Survey forms were adapted from the standard Water Vole Survey Form set out in the Water Vole Conservation Handbook (Strachan *et al*. 2011) (see Appendix I) to take into account the variation in field signs for the fossorial ecotype. Along with field sign counts, the site location and altitude was recorded using GPS (Garmin GPSmap76CSx), a site map hand-drawn and the following environmental variables also recorded: aspect of slope (using Recta DT200 compass), angle of slope, soil type, soil compaction/water logging/trampling, habitat type, neighbouring land use (recorded at an approximate visual distance of 100m of each site), NVC community/ dominant grass species, vegetation height, disturbance (e.g. digging by domestic dog, *Canis lupis familiaris*), presence of predators (*N. vison, Vulpes vulpes, Rattus norvegicus, Felis catus*).

Each PSU was surveyed twice during 2014, recording both pre-breeding (March-April) and post-breeding (Sept-Oct) field signs to establish temporal persistence of water vole populations at each site. Once field signs were counted a score of presence or absence was allocated to each site. Presence was further subdivided into three categories to give an overall idea of habitat suitability based on habitat usage by the water voles as indicated by field signs (see Table 3.1).

Table 3.1: Table of Presence/Absence Score for Water Vole Field Signs

Presence/absence Score	**Field Signs Present**	**Habitat Suitability**
0	No field signs recorded	Habitat unsuitable/never been occupied
1	Old burrows and/or tumuli	Habitat no longer suitable/temporary residence
2	Fresh signs of: burrows, tumuli, lawns/foraging remains, blocked tunnels, droppings	Habitat suitable to sustain water vole population
3	Fresh signs of: latrines, expelled nesting material (in addition to field signs in category 2)	Habitat suitable to sustain a breeding water vole population

3.1.3 -Presence of Predators

The main predator of the East End water voles is thought to be *V. vulpes* although other species like *R. norvegicus* and *F. catus* are also thought to be significant predators. Given the distance from water it was considered unlikely that *N. vison* would be present, which was also backed up with records from the Glasgow Natural History Society dating from 1962 which showed no records in the area. However, all scat and other predator signs like feeding remains were recorded and identified along the trapping transect.

3.1.4 - Disturbance

Disturbance was ranked on a scale of 1 to 3 with 1 being low disturbance and 3, high. Each site assessed for the following factors: trampling (human or livestock), noise (eg, traffic), frequency and intensity of human interference (e.g. grass cutting, use of public park), and the frequency of dug out burrows (e.g. by dog or fox).

3.1.5 - Dominant Grass Species

All grass species were recorded and identified to at least Genus level and their percentage cover and relative abundance ranked using the DOMIN scale (see Table 3.2) of the National Vegetation Classification system. The NVC system was developed in 1975 by the Nature Conservancy Council to produce a robust classification system which encompasses all the plant communities found across Britain which relates sward composition to other ecological factors (e.g. elevation, basal rock) (Rodwell, 2006). This system is also widely used by ecologists and land managers. Average sward height was recorded. Examples of sites can be seen in Fig 3.3 and 3.4.

Table 3.2: Percentage vegetation cover and DOMIN Scale

Domin Scale Number	Cover Percentage (%)
10	91–100%
9	76–90%
8	51–75%
7	34–50%
6	26–33%
5	11–25%
4	4–10%
3	<4% (many individuals)
2	<4% (several individuals)
1	<4% (few individuals)

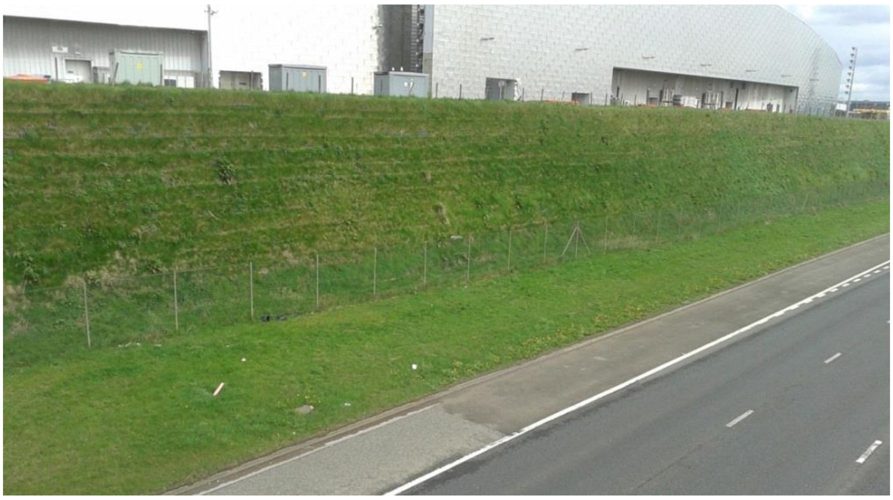

Fig 3.3: Glasgow Fort Green Wall with M8 in foreground (site 12.1)

Fig 3.4: Strone Road site viewed from Skerryvore Road (site 31.2)

3.2 Water Vole Field Signs

Water Vole Field Signs	**Description/Identification**
	*Burrow* Diameter 4-8cm. Wider than burrow is tall. Well-defined opening when in use Clipped grass around entrance (lawn). Fan-shaped soil mound outside (occasional). Identical in riparian and fossorial populations.
	*Tumulus* Flattened soil mound (diameter variable) Smaller than molehill. Can be singular but normally found in clusters. Cluster arrangement with each tumulus touching neighbouring one. More frequently recorded in European fossorial populations (Meylan, 1977).
	*Droppings* 8-12mm length. Circular in diameter with blunt ends Green when fresh. Dark brown when dried out. Identical in riparian and fossorial populations.

Water Vole Field Signs	Description/Identification
	*Feeding Station* Collection of vegetation foraging remains. Can be varying lengths but always cut at a 45° angle (Strachan & Moorhouse, 2006). Tend to be situated at base of a grass tussock. Diet will vary between populations due to differences in habitat.
	*Blocked Burrow Entrance* Burrow entrance blocked by water vole using a mixture of soil, shredded grass and moss. Riparian voles block tunnel entrances when river levels are high to avoid flooding (Strachan & Moorhouse, 2006). Blocked tunnels common in fossorial ecotype (Meylan, 1977). Common to riparian and fossorial populations.
	*Latrine* Piles of droppings “drummed” and scent-marked by females. Highly seasonal – occurs during the breeding season. Marks territory boundary for females (Strachan & Moorhouse, 2006). Arrangement may vary between riparian and fossorial populations

Water Vole Field Signs	**Description/Identification**
	*Expelled Nest Material* Shredded mixture of grasses forming nest which are pushed above ground by water vole when cleaning out burrow nest chamber. Can occur any time of year but more common after breeding season and in preparation for winter. Common to both riparian and fossorial populations.

3.3 - Abundance of water vole using capture-mark-recapture (CMR)

3.3.1 - Trapping Sites

Six sites were selected for CMR based on advice from GCC (Table 3.3) and results from the pre-breeding surveying effort under the premise that high levels of field signs may represent core colonies and could therefore be important for long-term water vole conservation in the area. There was one exception, Garthamlock Water Towers, where permission for site access was not granted until June 2014 and the abundance of field signs were unknown prior to this. However, trapping was unsuccessful at this site.

3.3.2 - Trapping Methodology

At each site population traps were set along a 100m transects with paired traps, each of a different type, set at 10m intervals, giving a total of 20 traps. Trapping protocol was based on guidelines from Gurnell & Flowerdew (2006) and trap positioning based on Telfer *et al*. (2003a) and advice from Dr Matthew Oliver, Dr Sandra Telfer and Emma Bryce (*pers. comm.*).

Table 3.3: Table of sites and dates for live trapping and CMR. Note the two trapping attempts at Cranhill Park: the first was aborted due to trap theft and interference.

Site Name	Trapping Dates	Landowner	GRID ref	Habitat Type	Disturbance
Avenue End Primary School	09/06/14-14/06/14	GCC	NS 649666	Park/garden: managed grassland	Low
Garthamlock Water Towers	30/06/14-04/07/14	Scottish Water	NS 653665	Park/garden: managed grassland	Low
Tillycairn Drive	28/07/14-01/08/14	GCC	NS 656667	Park/garden: managed grassland	Moderate
Cranhill Park	14/07/14-16/07/14 (1st attempt) 11/08/14-15/08/11 (2nd attempt)	GCC	NS 644655	Park/garden: managed grassland	High
Hogganfield Park	Did not take place – time constraints	GCC	NS 646673	Park/garden: managed reedbed	Moderate
Croftcroighns Park	Did not take place – time constraints	GCC	NS 644662	Park/garden: managed grassland	High

Each trapping session was conducted over a consecutive 5 day period with traps installed on the first day and left unset for approximately 24 hours to become "acclimatised". The paired traps at each trapping point were set at a 90º angle to an obvious field sign (e.g. burrow entrance) at a distance of 50-100cm depending on the terrain. Traps were numbered prior to use and each trapping point was marked by a numbered length of bamboo cane to easily allow each point to be ticked off on a check-sheet to ensure none were missed at each trap check.

Based on the trapping protocol from Strachan *et al*. (2011) each trap was provisioned with roughly 80-120g chopped carrots and fresh hay for bedding, with a handful of chopped apple placed at the entrance of the trap as bait (see Fig 3.5 and 3.6). Traps were secured in position by placing them in a well defined water vole run or beneath a grass tussock on flat ground. Once secured they were covered with tall grass/vegetation at the site to provide shelter, some degree of insulation and a visual barrier from predators and human interference.

Fig 3.5: Sherman trap in situ

Fig 3.6: Tube trap in situ

(Photographs courtesy of Stef Scott©)

Daily trap checks were conducted three times per day at set time intervals: 0500, 1300 and 2100. Frequent checks meant captured animals would never be left for a time period in excess of 8 hours and the trapping schedule was modified if necessary in order to ensure animal welfare and avoid fatalities e.g. prolonged spells of rain or colder than average overnight temperatures. Lactating females were also a significant concern hence the rigorous trapping schedule to prevent them being away from any young for a prolonged period. Traps were cleaned, re-provisioned and reset as necessary after each check. Once the 5 day trapping was complete, Sherman traps were autoclaved and Tube traps disinfected.

3.3.3 - Trap Types

Sherman Folding Traps (model XLK) (see Fig 3.7) are recommended by Strachan *et al*. (2011) for live trapping water voles, however, previous attempts using them proved unsuccessful at capturing the East End animals. Trap theft was also a serious concern for this section of the project given that a previous Masters student trapping in Cranhill Park had all 30 traps stolen and that for this study 4 out of the 6 sites were in urban parks (Stewart White, *pers comm*.).

Fig 3.7: Tube trap and Sherman trap with ruler for scale (30cm)

To attempt to overcome this, a trap prototype was developed by University of Glasgow technicians based on a simple design from a German trap (Rohrenfalle) used for trapping fossorial water voles and moles which could prove more suitable for the East End animals. The prototype, the Tube trap (see Fig 3.7), was constructed out of grey plastic plumbing pipe with one-way hinged doors at either end. It is essentially a scaled-up version of the German design with an increase in length and diameter to take into account the larger body size of British water voles compared to European fossorial populations. This simple design was hoped to deflect any interest and possible interference from members of the public. The Tube trap also has the benefit of being robust, light-weight and given its shape, can be easily slotted into the above ground runaways created by water voles making it secure and unlikely to move when positioned correctly. They did, however, prove to be attractive to slugs.

3.3.4 - Avenue End Pilot Study

A pilot study into trap preference was conducted for a 5-day consecutive period from the 9th – 14th June 2014 at Avenue End Primary School. This site was chosen because it is secure with no public access. Traps were set using the methodology previously described and in addition 2 Acorn™ 5210A Wildlife Trail Cameras with Standard Infrared and 2 Bushnell™ Trophy Cameras were positioned along the transect line to enable remote observation of water vole presence and above ground activity levels. It was hoped that by using these 2 methods along with field signs, trap shyness/avoidance could be investigated. However, trapping was unsuccessful.

3.3.5 Animal Handling and Marking

Animal handling protocol was followed from Strachan & Moorhouse (2006) with captured animals transferred into a pop-up garden waste container (Gardman™ Heavy Duty Polyethylene, height 58cm). The water vole was then encouraged into a cardboard tube (Pringles™) (see Fig 3.8) and the tube and contents weighed using digital scales (DIPSE PS-250). The tube was weighed at the start of each trap check and recorded for later use to calculate true body mass (see Fig 3.9). Once the animal was weighed the base of the tail was grasped to allow sexing, ageing (adults were classed as individuals over 100g) and the recording of body length ± 0.02cm (nose to base of tail) using a metal ruler, tail length, hind foot length and anal-genital gap.

Coat colour, presence/absence of ectoparasites (e.g.Gamasid mites and fleas) and general body condition were also noted. Once the animals were measured they were then marked for CMR using Passive Integrated Transponder (PIT) tags.

Fig 3.8: Juvenile water vole investigating the cardboard tube

AVID™ Single-use Sterile Syringes PIT tags were injected subcutaneously into the scruff between the shoulder blades. Once this procedure was carried out the animal was placed back in the garden container and monitored for 5 minutes. When recovered and active they were scanned using the AVID™ Mini-Tracker Microchip Scanner to ensure marking was successful and released back into the environment at the point of capture under a grass tussock for cover. If the success of the marking was in doubt then as backup a hair clip was taken and recorded (see Appendix III). This procedure was carried out under Home Office Licence and by the Licensee only. Direct handling was kept to a minimum to minimise stress. The level of stress of each individual was assessed after every marking procedure and logged for a later Home Office Report. Water voles have a powerful bite and bite-proof gauntlets were worn during handling.

3.3.6 - DNA Sample Collection

Ear biopsies for genetic analysis were to be collected from all captured individuals. This was not possible however, due to the difficulties of suitably restraining the animal while attempting the delicate procedure of taking an ear biopsy while lone working. Water voles are too large an animal to restrain using the scruff-holding method employed in smaller mammals like mice and is not recommended in Home Office training. Four ear biopsies were taken from dead animals (not fatalities from trapping) and have been stored in ethanol for future analysis which is out with the scope of this project.

Fig 3.9: Weighing a water vole using cardboard tube and digital scale

3.4 - Habitat Classification and Assessment

3.4.1 Phase 1 Habitat Type

Each of the trapping sites was classified by Phase 1 Habitat type and assessed in relation to water vole abundance. Each trapping transect was subject to the same methodology used for the presence/absence field sign surveys and the key environmental variables were recorded: altitude, aspect of slope, angle of slope, soil type, soil compaction/water logging/trampling, habitat type, neighbouring land use, NVC community/dominant grass species, vegetation height (average sward height), disturbance, presence of predators (*N. vison, V. vulpes, R. norvegicus, F. silvestris catus*).

3.4.2 National Vegetation Classification (NVC)

Quadrats (2m x 2m) were conducted at three points along the trapping transect, at transect point 1, 5 and 10. All plant species were recorded and identified to at least Genus level and their percentage cover and relative abundance ranked using the DOMIN scale (see Table 3.2) of the National Vegetation Classification system. Average sward height was recorded. Soil samples were taken at each quadrat point from a depth of 10-15cm and the pH analysed using Westminster Soil Testing Kit (see Appendix IV).

3.5 - Data Analysis

3.5.1 Presence/Absence Data

All presence/absence data was entered into Microsoft Excel with separate workbooks for the March/April (Spring) surveys and the September/October (Autumn) surveys (see Appendix II for example). Bar charts were created to show the percentage relative difference between occupied and unoccupied sites for Spring and Autumn surveys for the environmental variables investigated.

3.5.1a: Binary Logistic Regression

Minitab (Version 17.2) was used for multivariate analysis to investigate the incidence of water voles in response to the measured environmental variables: habitat type, vegetation type, vegetation height, disturbance levels, altitude, slope profile, slope aspect, distance from riparian habitat, distance from the M8 corridor, presence of predators and neighbouring land use. Logistic regression is a powerful tool for use in habitat preference studies with random sampling (Keating & Cherry, 2004). Where appropriate, environmental variables were categorised (e.g. habitat type defined by primary land use and management regime) and modelled using binary logistic regression with presence or absence as the response variable. Distance from water was measured as direct overland distances using Google Earth, as was distance from the M8. Distance from the M8 was excluded from habitat analysis because it might be a confounding factor with site isolation rather than an indicator of preferred habitat. Categorical variables with counts of less than 5 were excluded from the model in order to maximise the model's robustness. Data was log transformed. A backwards stepwise process of elimination of the variables from the full model was used to find the final model with the lowest AIC weight (Akaike's Information Criterion). The odds ratio output from the model allows for a quantitative comparison of the effect of each category relative to the other categories within the explanatory categorical variable. The odds ratio represents the constant effect of predicator x. An odds ratio of 1 equates to even odds, whereas, 1.10 represents a 10% increase in the effect.

3.5.1b Principle Component Analysis

The binary data for dominant grass species was analysed for Autumn surveys only. Spring surveys were excluded because of the difficulties in identifying old and dead grass from the previous year's growth. For PCA, the Microsoft Excel workbooks were saved as a separate csv (comma delimited) file and the data log(x+1) transformed to eliminate the effect of multiple zeros within the data set. The occurrence of dominant grass species was then analysed between occupied and unoccupied sites using the Bray-Curtis measure of similarity in SIMPER analysis.

This estimates the percentage similarity of dominant grass species in relation to water vole incidence giving their percentage contribution out of 100% for occupied and unoccupied sites. Occupied sites were further subdivided by level of water vole occupancy: peripheral use, feeding and breeding and analysed for relative similarity in dominant grass species using SIMPER.

3.5.2 - Trapping Data

3.5.2a Biometrics

Trapping data was entered in Microsoft Excel workbooks for each site. Excel was used to present the accumulation curves of number of captures per site and daily activity patterns graphically. The data was analysed using Minitab 17.2 and where appropriate, trapping variables were categorised, put into contingency tables and analysed using Chi-squared. Biometric data was analysed using regression analysis in Minitab17 and significant results presented in a fitted line plot. The relative abundance of field signs at each trapping site was also analysed this way.

3.5.2b CMR Data

Program MARK was used to model water vole population estimates at each site based on CMR data. Capture histories for each site were created in .inp files and opened in Program MARK. Although the populations sampled were technically open, a closed population with no migration or birth/deaths (and therefore a constant number of animals) was assumed for each site because of the short trapping timescale. Closed population "Huggins p and c" model was chosen where N, the population estimate, is a derived parameter based on the number of animals detected and assumes an equal probability of capture for all individuals. A closed model was chosen given the short trapping duration. The most parsimonious model was used in each case based on the model with the lowest AICc value against the highest AICc weight and lowest number of parameters. The goodness of fit was tested using the variance inflation factor ($\hat{c}$) and 120 simulations run for each model. Due to the use of linear trapping methodology in a non-linear habitat (because of time and money constraints) only relative abundance estimates can be calculated for the areas sampled rather than true population density. The length and width of each trapping site was measured using a 100m measuring tape and the total area of each grassland patch calculated in m^2.

4. RESULTS

4.1- Presence/Absence Field Sign Surveys: Water Vole Distribution

Sixty-five out of the total 68 sites were surveyed for water vole presence or absence over March/April and 62 sites re-visited between Sept/Oct. There were no suitable sites for field sign surveys for site number 2.1, 2.2 or 20.2 and these were excluded from both Spring and Autumn surveys. Site numbers 10.2, 18.1 and 28.1 were not repeat-surveyed because of livestock and access issues.

Water vole presence was recorded in 21 sites in March/April and 19 sites in Sept/Oct. The repeat surveys carried out in Autumn showed 21% of sites were occupied by persistent populations, 7% of sites were no longer occupied (sites 5.2, 7.1, 34.1, 34.2), 3% of sites had been newly colonised (sites 11.2 and 29.2) and 69% of sites remained empty. Two out of the four sites which were no longer occupied (sites 5.2 and 7.1) only one burrow was recorded per transect in the Spring surveys, whereas multiple field signs were recorded at 34.1 and 34.2 indicating a higher level of water vole habitat use. At the newly colonised sites one feeding station was recorded at 11.2 and at 29.2, one feeding station and 2 tumuli, indicating relatively low habitat use.

The pattern of water vole distribution across the Seven Lochs study area shows a clear division between the local authorities of Glasgow City Council and North Lanarkshire, with no sites within North Lanarkshire occupied by water voles. Occupied sites appear to be concentrated along the M8 corridor and nearby urban sites, although numerous sites within 50-150m of large water bodies e.g. Hogganfield Loch, Bishop Loch and Gartloch (sites 4.1, 4.2, 15.1, 21.2) and 50m of marsh/reed bed (site 11.2, 15.2, 22.2, 29.1, 29.2) were also occupied (see Fig 4.1a and Fig 4.1b).

Water vole occupation was linearly related to distance from the M8 with sites closer to the M8 more like to be occupied (Linear regression: Presence = 0.6288 – 0.0001distance from M8/m df= 1, 64, $F= 24.94$, $p=0.001$).

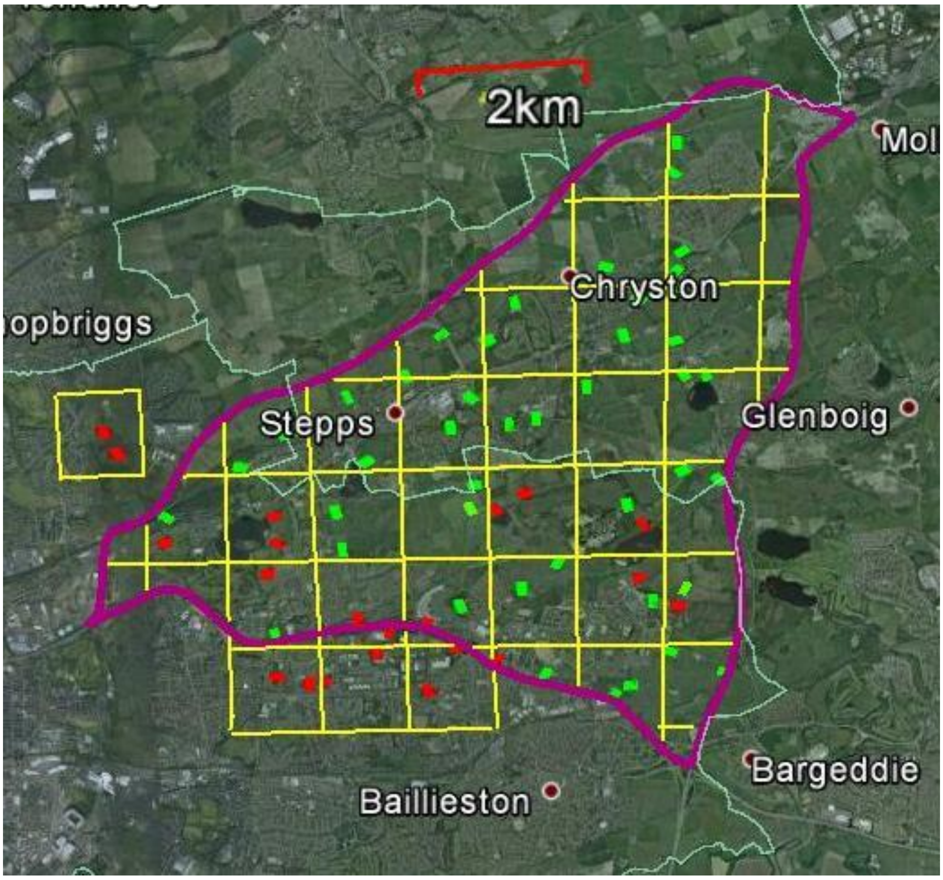

Fig 4.1a: Google satellite image showing Spring distribution of water vole presence (red) or absence (green) across study area. Council boundary (blue line) shows North Lanarkshire to the north and Glasgow to the south.

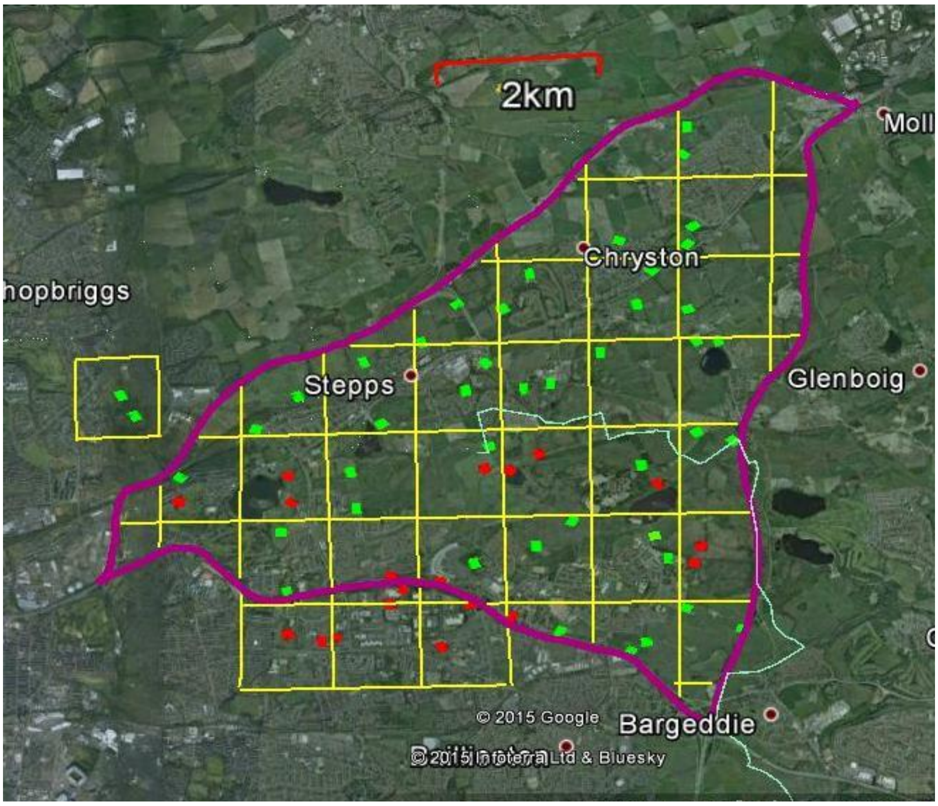

Fig 4.1b: Google satellite image showing Autumn distribution of water vole presence (red) or absence (green) across study area

4.1.1- Seasonality of Field Signs

The relative frequency of water vole field signs varied significantly by season (df 6, N=122 χ^2=13.962 p=0.03). Burrows, blocked tunnels, tumuli and droppings were equally likely to be recorded on both survey occasions. Feeding remains were twice as common in Autumn compared to Spring surveys. Latrines and nesting material were only recorded in Autumn surveys (see Fig 4.2).

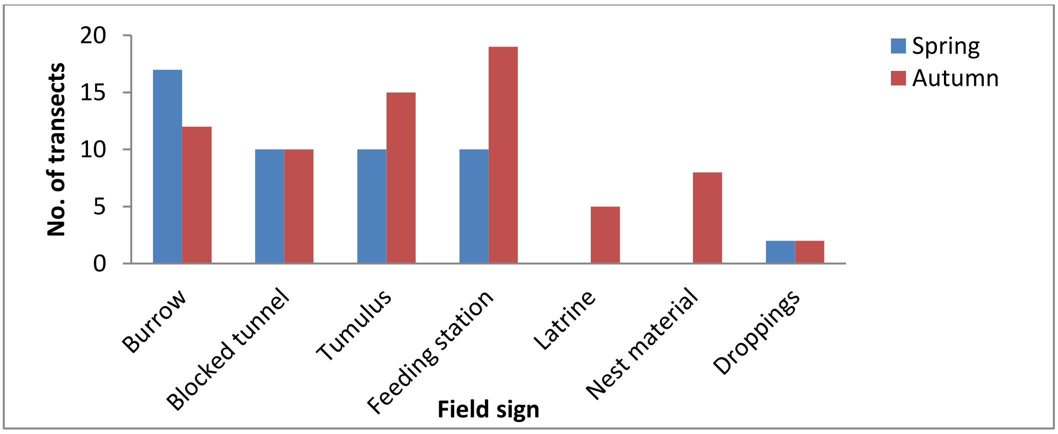

Fig 4.2: Relative frequency of water vole field signs between survey seasons

4.1.2a - Environmental Variables

Habitat Type

Water vole presence was recorded with the greatest frequency (16 out of 21 occupied sites in Spring surveys and 12 out of 19 occupied sites in Autumn surveys) across a range of urban grassland habitat types: park/garden, road verge, vacant land and industrial sites. Semi-natural and natural habitats which included permanent grassland, agricultural, open fields, rank grassland and marsh habitats were less frequently occupied by water voles (See Fig 4.3a and 4.3b). All four sites where water voles were detected in Spring but not in Autumn surveys were urban parkland. The newly colonised sites were semi-natural habitats, one being agricultural and the other permanent grassland.

Altitude, Aspect and Profile

The altitude of sites ranged from 65-121m above sea level with a mean of 88.9m ±11.1m. The majority of occupied sites (19 out of 21 sites in Spring surveys and 16 out of 19 sites in Autumn surveys) were in the 71-110m altitude range (see Fig 4.4a and 4.4b).

The mean profile of sites was 6.3º ±8.4º with a range of 0-50°. Flat or gently sloping sites (profile between 0-3°) were more frequently occupied by water voles compared to steeper sites (10 out of 21 sites in Spring surveys and 10 out of 19 sites in Autumn surveys). However, the majority of grassland sites surveyed were gently sloping (48 out of 65 sites). Only 17 out of the 65 sites had a profile of 10° or greater (10 sites with 10°, 2 sites with 16-20º, 3 sites with 21-25º, one site with a profile of 30º and the unique site, the Fort Green Wall, had a profile of 50º). Although there were fewer sites with a steeper profile, almost half of them were occupied by water voles in Spring surveys but this decreased to 38% in Autumn surveys (see Fig 4.5a and 4.5b). Aspect was subdivided by intercardinal points to be either north-, east-, south- or west-facing. A total of 23 sites were north, 18 south, 10 west and 2 east. Spring surveys recorded 8 north, 6 south and 5 west-facing sites as occupied. Five north, 8 south and 5 west-facing sites were occupied in Autumn surveys. Sites with a profile of 0° and therefore no slope aspect (a total of 12 sites), showed water vole occupation on two sites in Spring surveys and one site in Autumn surveys (see Fig 4.6a and 4.6b). The newly colonised sites recorded in Autumn surveys were both south-facing slopes whereas, sites which were no longer occupied in Autumn were flat or north-facing.

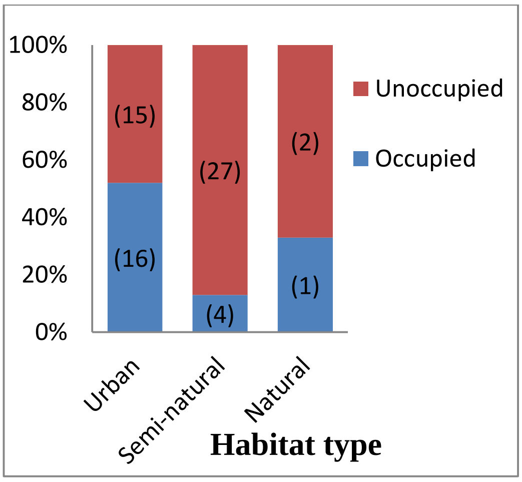

Fig 4.3a: Percentage occupied/unoccupied sites in Spring surveys by habitat type (number of sites indicated in brackets)

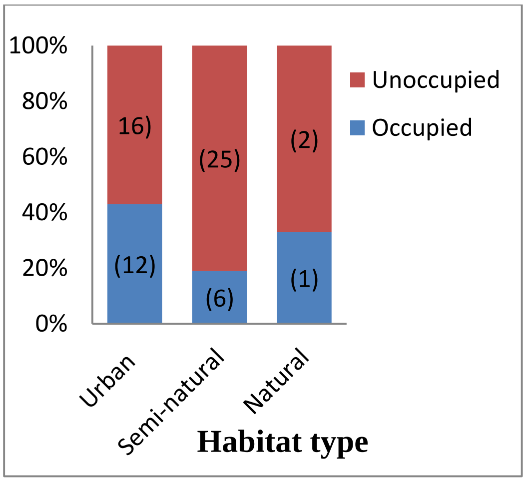

Fig 4.3b: Percentage occupied/unoccupied sites in Autumn surveys by habitat type (number of sites indicated in brackets)

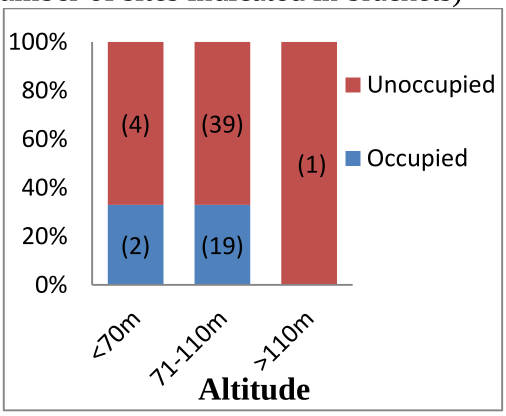

Fig 4.4a: Percentage occupied/unoccupied sites in Spring surveys by altitude (number of sites indicated in brackets)

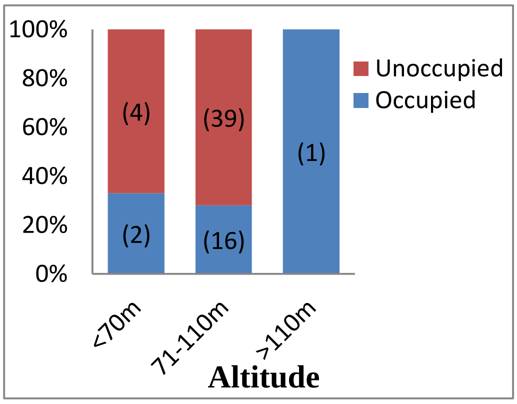

Fig 4.4b: Percentage occupied/unoccupied sites in Autumn surveys by altitude (number of sites indicated in brackets)

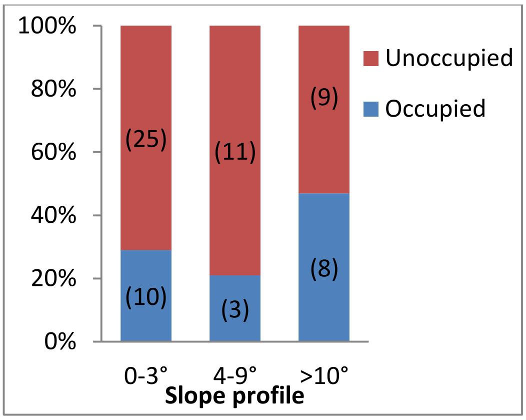

Fig 4.5a: Percentage occupied/unoccupied sites in Spring surveys by slope profile (number of sites indicated in brackets)

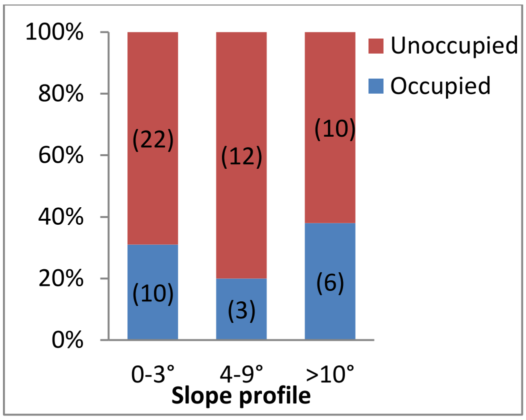

Fig 4.5b: Percentage occupied/unoccupied sites in Autumn surveys by slope profile (number of sites indicated in brackets)

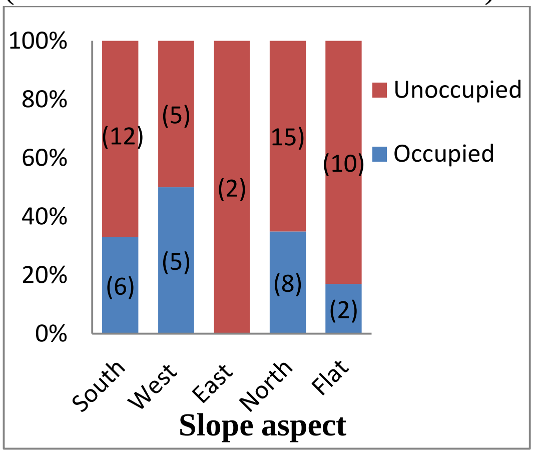

Fig 4.6a: Percentage occupied/unoccupied sites in Spring surveys by slope aspect (number of sites indicated in brackets)

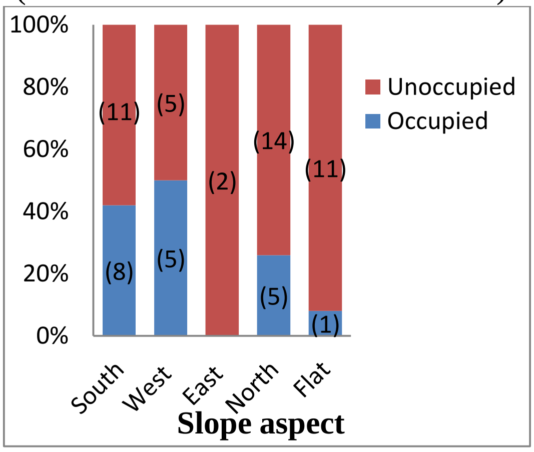

Fig 4.6b: Percentage occupied/unoccupied sites in Autumn surveys by slope aspect (number of sites indicated in brackets)

Vegetation type

Sites with a complex grass sward consisting of a mixture of short and tall grass species were most frequently occupied by water voles (12 out of 19 sites in Autumn surveys). This was based on Autumn surveys given the winter die back of vegetation for the Spring surveys. Water vole presence was also associated with the occurrence of rush species (e.g. *Juncus effusus*) (6 out of 19 sites). Sites with short grass (with or without rush), weeds (e.g. *Rubus fruticosus*), moss or heath were never occupied by water voles (see Fig 4.7). The four sites which were no longer occupied in Autumn were all dominated by tall grasses with/without rush. The newly colonised sites were tall grass with rush and grass mix with rush.

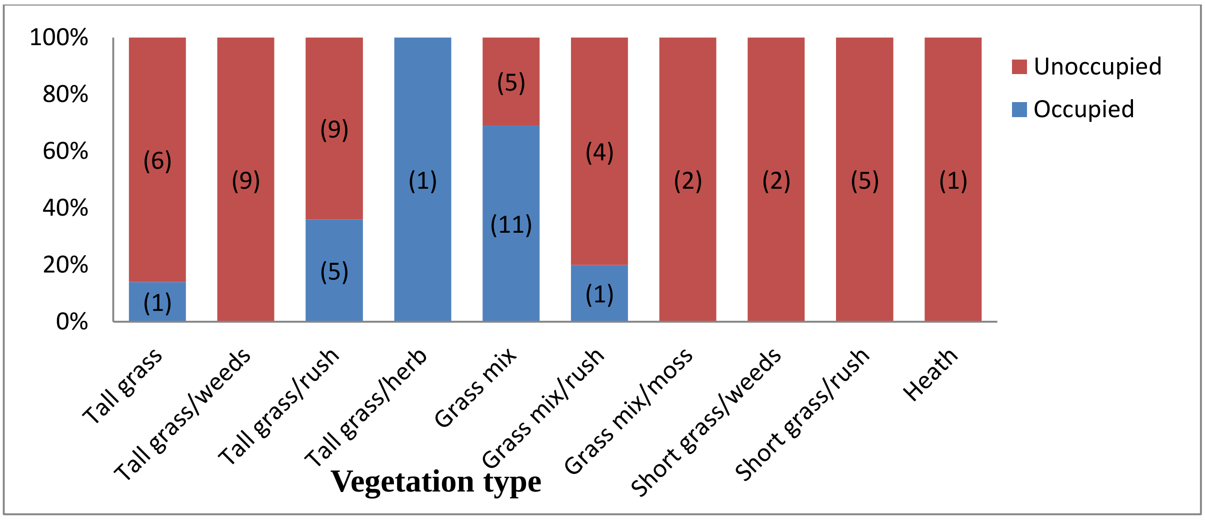

Fig 4.7: Percentage occupied/unoccupied sites by vegetation type (Autumn surveys only)

Average Sward Height

Seventeen out of the 19 occupied sites had an average sward height of greater than 26cm (see Fig 4.8). The mean sward height for occupied sites was 39.2cm ±15.5cm and ranged from 25-80cm for average sward height. The mean sward height for unoccupied sites was 43.4cm ±19.1cm and ranged from 5-80cm for average sward height. Spring survey vegetation height was not included because the vegetation was the previous year's growth and flattened after the winter.

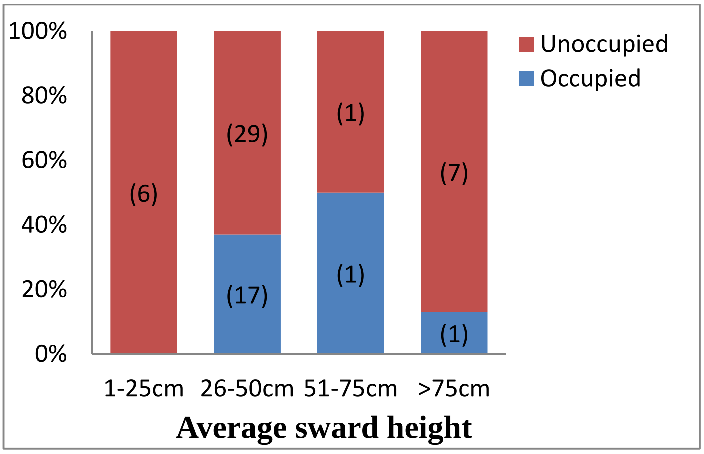

Fig 4.8: Vegetation height for Autumn occupied/unoccupied sites

Predator Signs

Sites were more frequently occupied by water voles when predators were absent (13 out of 21 sites in Spring surveys and 13 out of 19 sites in Autumn surveys) compared to when they were present. Foxes were the most commonly observed predator on survey sites (24 sites on both survey occasions), followed by domestic cats (7 sites in Spring and 5 in Autumn surveys) then the brown rat (one site in both surveys). American mink were never recorded. The presence of predators was detected on two out of the four sites which were no longer occupied in Autumn. No predators were detected on the two newly colonised sites.

Disturbance

Sites subject to moderate to high levels of disturbance were more frequently occupied than those with low disturbance (14 out of 21 sites in Spring surveys and 10 out of 19 sites in Autumn surveys) (see Fig 4.9a and 4.9b). Two-thirds of all sites were classed as low disturbance. Three out of the four sites which were no longer occupied were subject to moderate disturbance levels and both newly colonised sites were classed as low disturbance.

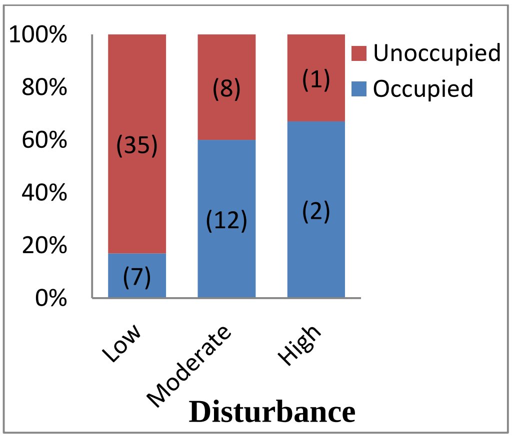

Fig 4.9a: Percentage occupied/unoccupied sites in Spring surveys by disturbance (number of sites indicated in brackets)

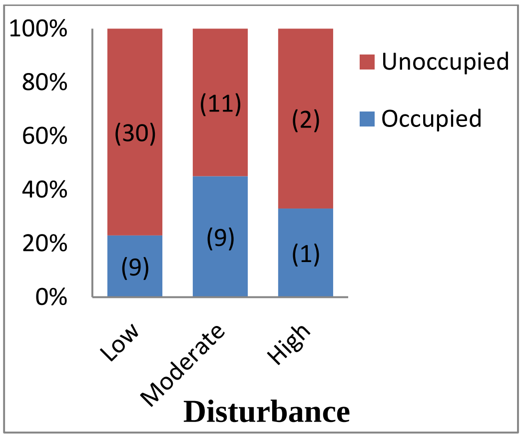

Fig 4.9b: Percentage occupied/unoccupied sites in Autumn surveys by disturbance (number of sites indicated in brackets)

Neighbouring Land Use

Table 4.1: Water voles presence/absence recorded in Spring and Autumn surveys according to neighbouring land use of sites

Neighbouring Land Use	Spring Surveys		Autumn Surveys	
	Present	***Absent***	***Present***	***Absent***
Housing	3	12	2	13
Park/garden	6	2	3	5
Marsh	3	4	5	2
Woodland	2	4	1	5
Agricultural	0	8	1	6
Industrial	1	7	1	7
LNR	0	1	0	1
Road verge	6	7	6	4
Total	21	45	19	43

The neighbouring land use of occupied sites was most commonly urban including road verges and parkland (16 out of 21 sites in Spring surveys and 12 out of 19 sites in Autumn surveys) compared to more natural land use (e.g. broadleaf woodland, agricultural, marsh). Three out of the four sites which were no longer occupied in Autumn had parkland as their neighbouring land use and one site had neighbouring broadleaf woodland. The two newly colonised sites were located next to marsh (Table 4.1).

Distance From Riparian Habitat

The distance of sites from suitable riparian water vole habitat (e.g. pond, river, marsh, reedbed) was measured as the minimum direct overland distance using Google Earth. Occupied sites ranged from 50-1148m from riparian habitat with a median distance of 162m (see Fig 4.10). Unoccupied sites ranged from 45-816m from riparian habitat with a median distance of 570m. The sites which were no longer occupied in Autumn were 50, 52, 161 and 816m from riparian habitat. The colonised sites were 50m and 189m from riparian habitat.

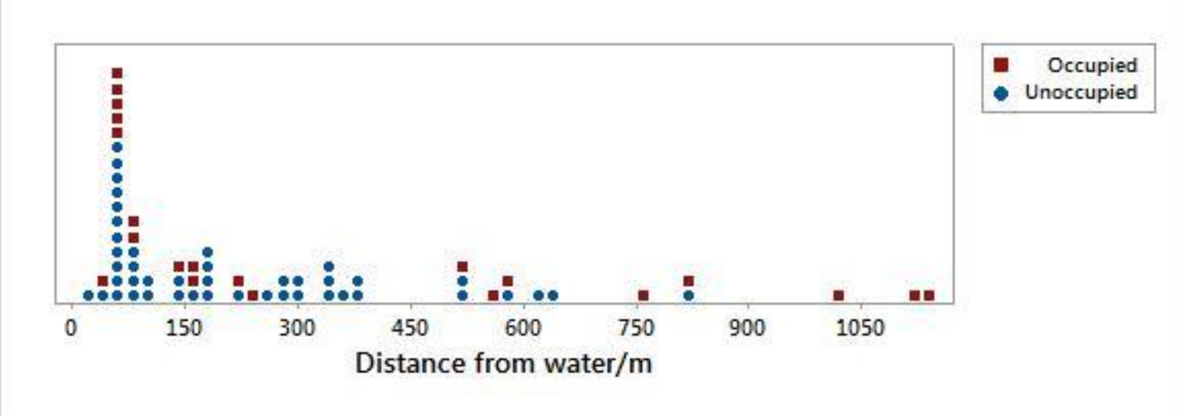

Fig 4.10: Occupied/unoccupied sites by distance from riparian habitat

This study shows that 43% of sites occupied by water voles were within 0-150m of suitable riparian habitat and 38% when sites were re-surveyed in Autumn. Sites at > 550m accounted for 33% of occupied sites and these colonies persisted to Autumn surveys. Twenty-three percent of occupied sites were at the intermediate distance of 151-550m. When field signs were subdivided by the level of water vole habitat use based on Autumn surveys, a total of nine sites supported breeding colonies, six of which were >550m from riparian habitat (site distances: 50, 336, 518, 567, 667, 754, 1012, 1126, 1148m). The other breeding colonies were Hogganfield Park grassland which was adjacent to reed beds (50m), the Fort Green Wall (336m from Auchinlea pond where water voles were absent) and the Queenslie M8 embankment site (518m). Feeding signs were recorded at eight out of the nine sites between 0-150m of riparian habitat and one site at the intermediate distance of 151-550m. Historic use was recorded on one site which was 0-150m from riparian habitat (site 29.1).

Distance from M8 Corridor

The distance of sites from the M8 corridor was measured as the minimum direct overland distance using Google Earth. Occupied sites ranged from 5-2600m from the

M8 corridor with a median distance of 520m. Unoccupied sites ranged from 80-7050m from the M8 corridor with a median distance of 2940m (see Fig 4.11). The four sites which were no longer occupied in Autumn were 56, 707, 897, 1520m distant from the M8 corridor. The colonised sites were 1840m and 1940m distant.

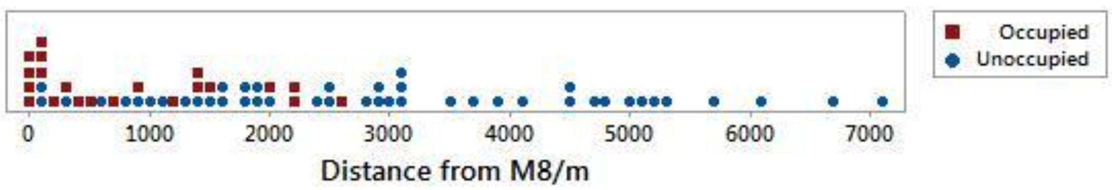

Fig 4.11: Occupied/unoccupied sites by distance from the M8 corridor

This study shows that 62% of all occupied sites were within 1km of the M8 corridor, decreasing to 53% when sites were re-surveyed in Autumn. When field signs were subdivided by the level of water vole habitat use, all nine sites supporting breeding colonies were within 520m of the M8 corridor (site distances: 5, 10, 10, 50, 60, 195, 300, 390 and 520m).

4.1.2b – Binary Logistic Regression: Environmental Variables

Binary logistic regression was used to examine the relationship between water vole presence and the environmental variables. Continuous explanatory variables were altitude, vegetation height and distance from the M8. Non-continuous factors were grouped into categorical variables (see Table 4.2). Habitat categories were grouped by management regime (e.g. grass cutting) and primary land use. Slope profile was split into three categories (flat or gently sloping, moderate slope and steeper slope). Slope aspect was split into five categories based on intercardinal direction (flat, north-, south-, east, or west-facing). Disturbance was split into two categories: low or moderate/high (the three sites classed as high disturbance were included in this category). The categories for distance from suitable riparian habitat were chosen to test whether the likelihood of water voles occupying a site decreased with increasing distance away from riparian habitat and were based on previous studies of water vole movements. The first category, 0-150m was selected based on the average territory size of 30-150m (Strachan & Moorhouse, 2006) and Moorhouse & MacDonald's (2005) radio-tracking study which measured water vole weekly ranges as 51.7-57.6m and 94.4-102.2m for females and males respectively. It was assumed that water voles occupying sites at distances <150m from riparian habitat were highly likely to be riparian populations as individuals would be able to move freely between riparian and grassland habitat. The intermediate distance category was classed as 151-550m from riparian habitat. The third category, >550m, was selected based on Telfer *et al*. (2001) four year study on fragmented, rural water vole metapopulations where colonies occupied a mean riparian length of 550-980m. The distance of 550m was taken as the maximum cut off point beyond which a water vole colony would no longer be able to function as a riparian population in the highly fragmented, urban environment of the East End (site distances from riparian habitat: 567, 667, 754, 816, 1012, 1126 and 1148m). Neighbouring land use was categorised based on the primary land use of neighbouring land at 100m distant. Season was categorised as changed (sites occupied/unoccupied in Autumn surveys) or unchanged (the site remained either occupied or unoccupied in Autumn surveys).

The variables slope profile, slope aspect and season were excluded from the regression model analysis because of multiple counts below 5. It was highly likely that vegetation type is a variable which influences site occupation, particularly given that water voles were never found in sites with short grass, but the zero counts meant it had to be excluded from analysis. Altitude was also excluded from analysis given the narrow range of 65-111m as water voles have previously been recorded over a wide altitudinal range (Saucy, 1994). Distance from the M8 corridor was also excluded given it is not a useful indicator of preferred habitat for water voles and the bias towards sites to the north of the trunk road (58 sites to the north and 7 sites south).

The model with the lowest AIC weight was selected using a backward stepwise removal of variables from the full model. The reference level for the categorical variables were set as park for habitat type, low for disturbance level and 0-150m for distance from riparian habitat and because of this their co-efficient equals 0. The model selected indicated that habitat type and distance from riparian habitat are important environmental variables which influence the occurrence of water voles in the East End (Regression P(1)= exp(Y')/(1 + exp(Y') df= 4, 64 p=0.001) (Y' = 0.656 + 0.0 Habitat category_Park - 2.509 Habitat category_Rank grassland - 0.796 Habitat category_Road verge + 1.295 Distance category_>550m + 0.0 Distance category_0-150m - 0.747 Distance category_151-550m). Hosmer-Lemeshaw goodness of fit test indicates that the fit of the model is good (df 5, x^2= 3.79, p=0.58).

Table 4.2: Binary Logistic Regression: Categorical Variables

Environmental Variable	Category	Presence (No. of sites)	Absence (No. of sites)
Habitat type	**Park** (includes parks, golf courses, playing fields, amenity grassland – high management)	9	5
	Road (includes road verge and industrial – moderate management)	7	8
	Rank (includes field verge, permanent grassland, vacant and - low management)	5	32
Slope profile	**Flat or gently sloping** (0-3°)	10	25
	Moderate slope (4-9°)	3	11
	Steeper slope (≥10°)	8	9
Slope aspect	**Flat**	2	10
	North-facing	8	15
	East-facing	0	2
	South-facing	6	12
	West-facing	5	5
Predator presence	**Present**	8	23
	Absent	13	21
Disturbance	**Low**	7	35
	Moderate (includes Moderate and High)	14	9
Distance from riparian habitat	**0-150m**	9	21
	151-550m	5	19
	>550m	7	5
Neighbouring land use	**Park** (includes parkland and housing)	9	14
	Road (includes road verges and industrial)	7	14
	Natural (includes marsh, LNRs, woodland, agricultural)	5	17
Vegetation type	**Tall grass** (includes tall grass with rush/weeds/herb)	7	24
	Mix grass (includes mix grass with rush/weeds/moss)	12	11
	Short grass (includes short grass with rush/weeds/heath)	0	8

Season	**Unchanged** (remaining occupied or unoccupied) **Changed** (occupied/unoccupied)	15 4	41 2

Habitat type was a strong predictor of water vole occurrence (p=0.002) with parks and road verges being the favoured habitat type over rank grassland (Table 4.3). The odds ratios show that parkland and road verges were more likely to be occupied compared to rank grassland which was rarely occupied (0.09 [95% Cl 0.02-0.4]). Road verges were five times more likely (5.05 [95% Cl 1.17- 21.81]) to be occupied by water voles compared to rank grassland. There was no significant difference in the likelihood of occupation between road verges and parks.

The likelihood of site occupation did not decrease with increasing distance from riparian habitat (p=0.06) indicating that the distribution of water voles in the East End was not dependent on close proximity to riparian habitat. The riparian distance category odds ratio reveal there was no difference in the probability of a site being occupied between the distance categories 0-150m and 550m but that water vole occurrence was less likely at the intermediate distance of 152-550m (Table 4.3). This indicates that the East End water vole populations are split into those which exist in close proximity to riparian habitat and those which exist independently of it.

Table 4.3: Binary logistic regression model

Environmental variable	**Presence (n=21)**	**Absence (n=45)**	**Co-efficient (SE)**	**Odds Ratio (95% confidence limits)**	**P value**
Habitat category: Park Road Rank	 9 7 5	 5 8 32	 0 -0.796 (0.845) -2.509 (0.804)	Rank vs Park: 0.08 (0.02, 0.39) Road vs Park: 0.45 (0.09, 2.36) Road vs Rank: 5.54 (1.25, 24.56)	0.002
Distance category: 0-150m 151-550m >550m	 9 5 7	 21 19 5	 0 -0.747 (0.753) 1.295 (0.834)	 0-150m vs >550m: 0.27 (0.05, 1.40) 151-550m vs >550m: 0.13 (0.02, 0.77) 151-550m vs 0-150m: 0.47 (0.11, 2.07)	0.06
AIC weight	71.34				
Regression model:	P= 0.001	R^2=25.0	DF=4		

4.1.3 - Principle Component Analysis: Similarity Percentages (SIMPER)

Dominant Grass Species

All sites were classed as B2 Neutral Grassland (JNCC, 2014). Occupied sites had an average of 27% similarity in dominant grass species and unoccupied sites, an average similarity of 20%. Velvet grasses *Holcus mollis* and *H. lanatus* were the dominant species on 43% of occupied sites. The average similarity between occupied sites by *Holcus* species (ave. similarity = 11.54) was higher compared to unoccupied sites (ave. similarity = 2.89) (Table 4.4). These grasses were also the dominant species on 67% of all breeding sites. *Holcus* species were found most frequently on flat or gently sloping sites (<5º). The rush *Juncus effusus* also showed a high average abundance on occupied sites as did the creeping thistle, *Cirsium arvense*. *J. effusus* was found most frequently on gently sloping sites with a north-facing aspect. *Holcus* species were also the dominant grass species found on 6 out of the 9 sites where breeding signs were recorded.

Agrostis species and tufted hair-grass, *Descampsia cespitosa,* were the dominant grass species in 52% of sites where water voles were absent. *Agrostis* species and *D. cespitosa* both occurred on gently sloping sites but *D. cespitosa* was more frequently recorded on north-facing sites compared to *Agrostis*. False oat-grass, *Arrhenatherum elatius,* and weeds like nettle, *Urtica diocia,* were also more common. It is worth noting, however, that *A. elatius* was recorded at water vole feeding stations at the Garthamlock Water Tower site during trapping and *Agrostis* was recorded at Tillycairn. Cocksfoot, *Dactylis glomerata*, was the only dominant grass species found in equal abundance on sites irrespective of water vole occupation. It also occurred on gently sloping sites.

Table 4.4: Percentage similarity of dominant grass species between occupied and unoccupied sites (PCA – SIMPER)

Grass Species	***Occupied sites*** **Ave Similarity**	***Occupied sites*** **% Contribution**	***Unoccupied sites*** **Ave Similarity**	***Unoccupied sites*** **% Contribution**
H. mollis/ lanatus	11.54	43	2.89	14
J. effuses	5.01	19	1.44	7
Agrostis	3.25	12	5.82	29
D. glomerata	2.98	11	2.31	11
D. cespitosa	1.99	7	4.59	23
A.elatius	0	0	1.69	8

4.2.1 - Trapping summary

Site

A total of 48 individual water voles were captured over the five trapping sessions. No individuals were trapped at Avenue End or Garthamlock Water Towers and both of these sites have been excluded from the *Trapping* section results (see *Methods 3.3.4* for an account of Avenue End and Garthamlock). Cranhill accounts for the majority of captures, making up 81% of the total (54% for Cranhill 1 and 27% for Cranhill 2).

Sex and Age

Of the total individuals caught 44 out of the 48 were identified as adults and could be sexed. There was an unequal sex ratio with 29 females and 15 males (df 1, N= 44 $\chi^2 = 3.84$ p= 0.05 using Yates correction). Only 4 juveniles were trapped, 2 males and 2 females, all of which were at Tillycairn. One Cranhill male died due to stress on handling at the first trapping session and will hereafter be excluded from analysis except for the *Biometrics* section.

4.2.1a - Trapping Effort

Trapping was non-uniform between the three trapping sessions. Cranhill 1 was cut short due to trap theft and verbal threats while conducting field work and concerns about further wildlife crime. Cranhill 2 had two trap checks missed, one because of a overnight spell of prolonged, heavy rain and the other because of illness. Tillycairn was the only site where there was no interference. The number of traps set at each session also varied because of five Sherman traps and two Tube traps being stolen at Cranhill 1 (Table 4.5).

Table 4.5: Trap hours across the three trapping sessions

Site	Number of Traps	Number of Trapping Hours	Trap Hours
Cranhill 1	26	28	728
Cranhill 2	20	30	600
Tillycairn	20	48	960

4.2.1b - Accumulation of captures

The accumulation curves for both Cranhill 2 and Tillycairn show a steady increase in new captures over the duration of the trapping session, whereas Cranhill 1 shows a far steeper increase with no sign of reaching an asymptote (Fig 4.12). The trapping took place over five days and the accumulation curves indicate that this time scale was adequate to sample and mark a substantial proportion of water voles at Cranhill 2 and Tillycairn.

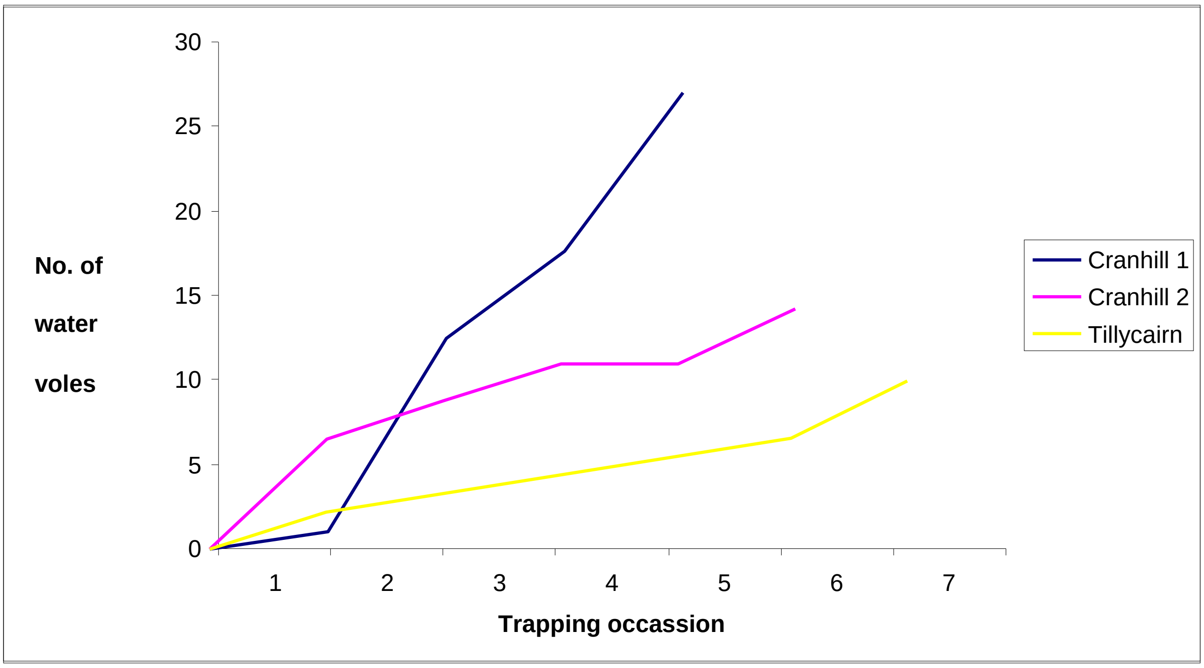

Fig 4.12: Accumulation curve of captures per trapping session

4.2.1c - Sex Ratios

The sex ratio of males to females (adults only) at each site was 2:3 at Cranhill 1, 2:11 at Cranhill 2 and 2:3 at Tillycairn (see Table 4.6). When juveniles are included, there was no significant difference between numbers of males and females across all three trapping sessions (df 2, N= 47 χ^2= 2.844 p= 0.241). Cranhill 2 differs from the 2:3 sex ratio of the other sites but was non-significant (df 1, N= 13 χ^2= 2.337 p= 0.126 using Yates correction).

Table 4.6: Number of males, females and juveniles per site

	Cranhill 1	**Cranhill 2**	**Tillycairn**
Male	10	2	2
Female	15	11	3
Juvenile	0	0	4
Total	**25**	**13**	**9**

4.2.1d - Diel Activity Pattern: Day vs Night Capture

Taking into account new individuals and recaptures, a total of 59 successful captures occurred across the three trapping sessions. The trapping schedule was such that the 0500 dawn check sampled individuals active during twilight/darkness and the 1300 check sampled those active during daylight hours. The time between each trap check was equal (8 hours) and although there were differences in trap hours between sites, Tillycairn and Cranhill 2 were balanced between day and night captures. At Tillycairn, night captures made up 47% of the total and day captures, 53% and was non-significant (df 1, N=15 χ^2= 0.033 p=0.855 with Yates correction). Cranhill 2 had a significant amount of day captures compared to night (df 1, N= 15 χ^2= 4.266 p= 0.039 with Yates correction). Whilst not directly comparable, there was a greater number of day captures at Cranhill1 (17 day and 12 night captures) but one extra daylight trapping occasion.

Camera traps at Avenue End Primary site set between the 22nd-25th of June photographed water vole activity starting at 0513, 0521, 0532 and 0526 on each consecutive day (see Fig 4.13). The camera traps were triggered during the night but the photographs were overexposed and the animal responsible could not be identified.

These night shots made up 17 out the total of 363 taken and the greatest activity was recorded between 0513 and 2210 (earliest and latest times).

Fig 4.13: Water vole at Avenue End. Traps were set at ground level because of the density of vegetation (taken 24th June at 0856)

Sex

Time of capture was analysed by sex across the three trapping sessions and proved to be significant with females more likely to be trapped during daylight hours (df 1, N= 59 χ^2= 5.622 p= 0.018) (See Fig 4.14). Males were more likely in night captures at Cranhill 1, whereas only daylight captures were recorded for males at Cranhill 2. Females were more likely to be trapped during daylight across all sites. However, if the unbalanced Cranhill 1 session is excluded, then the result is non-significant (df 1, N=31 χ^2= 0.136 p= 0.713).

4.2.2 - Trap type

4.2.2a Avenue End Pilot Study

No water voles were trapped during the pilot study conducted at Avenue End Primary (9-14th June). Field sign surveys showed that there were low levels of water vole activity (no droppings, latrines or fresh tumuli) on the site and this was also confirmed by footage from the camera traps which were positioned along the trapping transect with no photographs of water voles during the five days. There was some concern over predation by domestic cats and its effect on the Avenue End population as in previous years up to eight water voles could be witnessed feeding on the slope (Geraldine O'Donnell, *pers comm*). However, camera traps were left in situ until the 25th of June and 363 images were taken on a single camera trap from the 22nd-25th showing there was at least one highly active water vole on site.

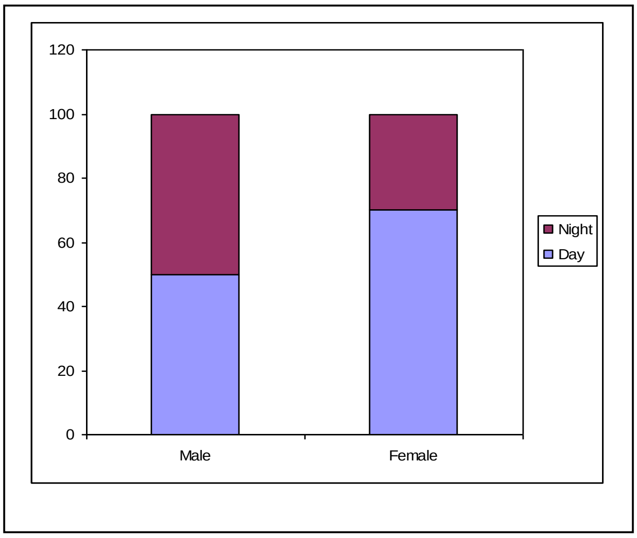

Fig 4.14: Percentage difference between day and night captures of males and females (Cranhill 1 excluded)

4.4.2b - Trap Preference

There was a significant preference for Sherman traps over Tube traps (df 1, N= 59 χ^2= 7.475 p=0.006). Sherman traps accounted for 68% of total captures. Sherman traps captured individuals with a mean body length of 148mm (range: 106-179mm) and Tube traps captured individuals with a mean body length of 144mm (range: 111-164mm). Mass also shows little difference between trap types: the mean for Sherman's being 108.3g (range 38.4-221.7g) and Tube trap mean being 92.6g (range 38.6-183.5g).The greater trapping success of Sherman traps was similar between sites (df 2, N= 59 χ^2 = 0.288 p= 0.866) with no preference evident between males, females or juveniles (df 2, N= 59 χ^2 = 0.274 p= 0.872).

4.2.3 - Biometrics

Mass

The mean body mass of the water voles sampled was 108.48g ± 35.6g (Table 4.7). Males were larger than females (mean mass 113g and 105.5g respectively) and adults were larger than juveniles (mean mass 114.5g and 42.4g respectively). Similar masses were recorded between males and females (GLM df 1,1,45 F=1.71 p=0.198). Site was non-significant when age was factored into the model (GLM df 1,1,45 F=0.34 p=0.562).

The presence of ecto-parasites had no effect on mass and there was no interaction between age and presence/absence of ecto-parasites (GLM df 1,1,45 F=21.89 p=0.001 (age) F=1.12 p=0.296 (ectoparasites)). Similarly, coat colour had no bearing on body mass.

Table 4.7: Water vole biometrics with mean, range and standard deviation

	Mean	**Range (Minimum – maximium)**	**Standard Deviation(SD)**
Mass (g)	108.5	38.4 – 221.7	35.6
Body length (mm)	144.4	106 - 179	15.9
Tail length (mm)	89.6	59 - 116	12.4
Hind foot length (mm)	29.5	24 - 33	1.95

Body Length

The mean body length was 144.4mm ± 15.9mm (see Table 4.7). Males had a longer body length than females (mean body length 151.1mm and 145.7mm respectively) but was non-significant. There was no significant variation in body length between sites

(GLM df 1,45 F=0.05 p=0.828) or as a result of coat colour or presence/absence of ecto-parasites although water voles with a brown coat colour do have a slightly longer body length (146.6mm) compared to black (144.2mm) and intermediate coat colours (141.6mm).

Mass and Body Length

Body mass and body length are linearly related (Regression equation: Mass (g) = -139.9 + 1.721 Body length (mm) df 1,45 F=66.87 p=0.001) (see Fig 4.15) although the majority of data points are clustered around the means for both body length and mass. The four Tillycairn juveniles can be clearly picked out at the start point of the regression line. Outlying points above the upper 95% CL represent four large males and one lactating female. Two individuals with poor body condition can be seen below the lower 95% CL (body length=139mm, 174mm; mass=51g, 103g respectively).

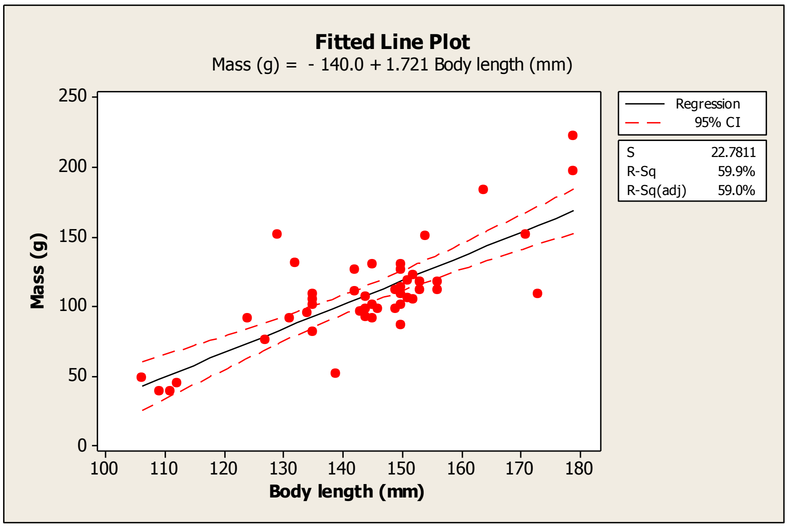

Fig 4.15: Regression analysis of body mass against body length

Tail Length

The mean tail length of water voles sampled was 89.6mm ± 12.4mm (Table 4.7).

Males show the same non-significant pattern in their biometrics when compared to females (GLM df 1,45 F=2.84 p= 0.099), having a slightly longer tail length (95.7mm and 90.2mm respectively). Site had no effect on tail length (GLM df 1,45 F=0.12 p=0.729). Ecto-parasite presence or absence was excluded from this analysis as mass and body length were considered more likely indicators of overall health compared to tail length.

Mass and Tail Length

Tail length and body mass are linearly related (Regression equation: Mass (g) = -103.3 + 2.356 Tail length (mm) df 1,45 F= 97.29 p=0.001) and tail length appears to be a more accurate predictor of water vole mass compared to body length (see Fig 4.16).

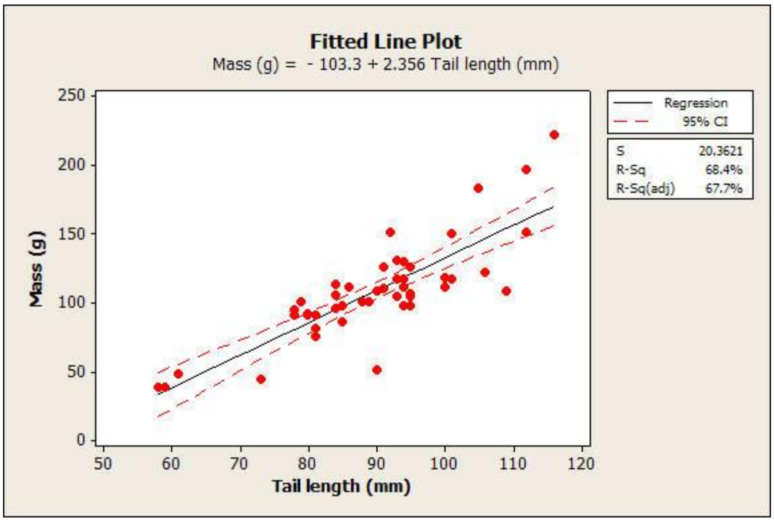

Fig 4.16: Regression analysis of mass and tail length

4.2.4 - Capture-Mark-Recapture Data

4.2.4a - Mark-recapture Summary

Out of the 48 water voles marked with PIT tags there were 12 recaptures in total. All recaptures were female except for one male trapped at Tillycairn. Only two animals were recaptured on multiple occassions: one female from Cranhill 1 was recaptured twice and another female from Tillycairn was re-trapped four times. Only adults were recaptured; no juveniles. All recaptures were at the same trapping point but trap type varied. No other small mammal species were trapped.

4.2.4b - Population Modelling and Population Estimates

The model with the maximum likelihood for Cranhill 1 showed there was variation in capture probability(p) over time for both the group as a whole and a difference in capture probability between males and females, but that recapture probability(c) was constant (Table 4.8). Derived population estimates for the total group are 78 (95% Cl 41-197). Simulations of ĉ show a good fit between observed and predicted estimates (see Fig 4.17).

Cranhill 2 derived population estimates are 42 (95% Cl 20-141) for the total population. Both capture and recapture probability were constant over time which was in marked contrast to Cranhill 1 but this could do down to the greater number of trapping hours at Cranhill 2. The goodness of fit of the model is questionable given the ĉ simulations output but this is probably down to the small sample size (see Fig 4.18).

Table 4.8: Table of population estimates and model output from Program MARK

Trapping session	**Model used**	**Population estimate (Ñ)**	**S.E.**	**Lower 95% CL**	**Upper 95% CL**	**No. of parameters**	**AICc**
Cranhill 1	{c(.)=p(t)}	78	35	41	197	4	90.99
Cranhill 2	{c(.)=p(.)}	42	26	20	141	1	58.16
Tillycairn	{c(.)=p(.)}	6	0.85	5	10	1	40.80

The sample size for Tillycairn was the smallest out of the three trapping sessions but it did have the greatest number of trapping hours. Both capture and recapture probability were constant over time and the derived population estimate was 6. This may seem unusual given 9 animals were trapped but 95% confidence limits are between 5 and 10 and this model indicates that there was a high likelihood that all the animals in the trapping area were marked during the study. It is possible that low sample size is also a factor but the abundance curve showed saturation was reached very quickly during the trapping period and ĉ simulations show a strong goodness of fit between observed and predicted values (see Fig 4.19). There is an increase in individuals towards the end of the trapping session but this is accounted for by the three newly emerged juveniles caught on the final occasion.

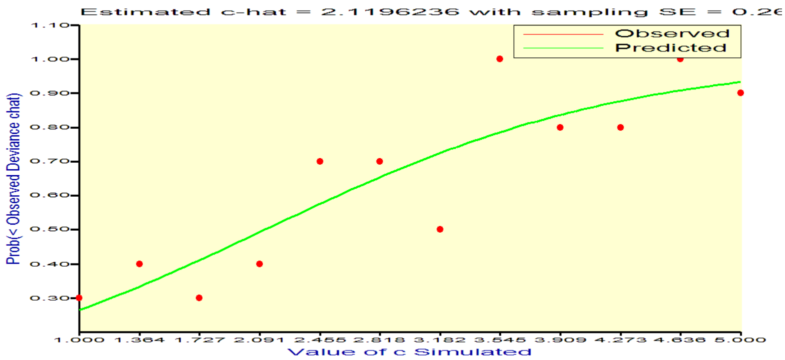

Fig 4.17: Simulations of ĉ for total population for Cranhill 1

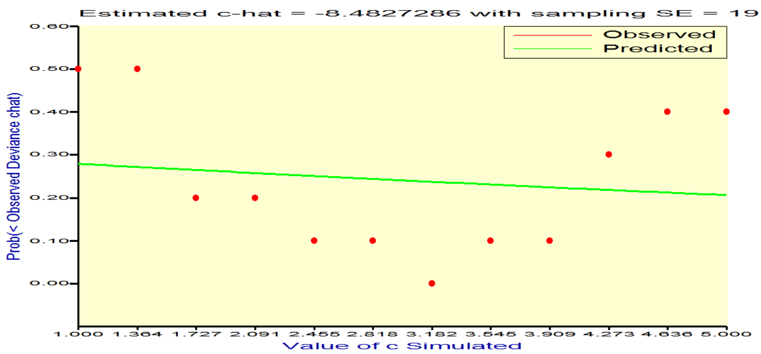

Fig 4.18: Simulations of ĉ for total population for Cranhill 2

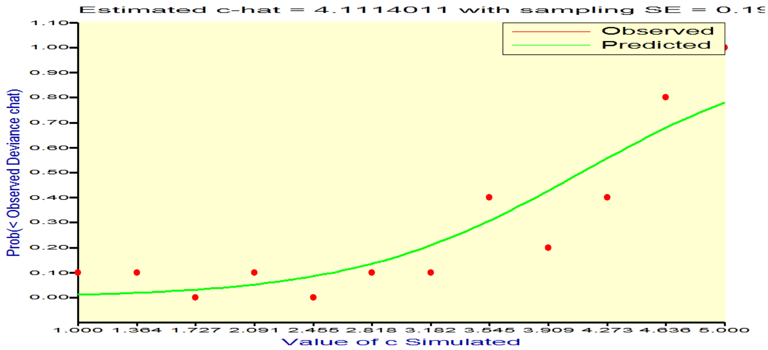

Fig 4.19: Simulations of ĉ for total population for Tillycairn

4.2.4c - Population Density

The landscape features of each trapping site varied. Tillycairn comprised of a long strip of grassland running along an area of banking (slope profile 10°) beside amenity grassland and housing; Cranhill was a roughly circular area of gently sloping grassland (slope profile 3°) within a city park. The area of each trapping site was measured and the population densities per hectare calculated by multiplying the total area by the population estimates from modelling.

Table 4.9: Population density estimates per hectare from Huggins p and c models

Site	**Area of habitat patch (m^2)**	**Population estimate from Huggins p and c model per hectare (95% Cl)**
Cranhill 1	5000	156 (82-394)
Cranhill 2	5000	84 (40-282)
Tillycairn	1500	40 (33-67)

4.2.5 - Relative Abundance of Field Signs: Trapping Sites

The relative abundance of latrines and the number of water voles captured during trapping sessions were linearly related (Regression equation: Latrine= 0.389 + 0.768

water voles, df 1,29 T=2.9, p=0.008) (Fig 4.20). The relative abundance of feeding stations was also linearly related (Regression equation: Feeding station= 1.82 + 0.897 water voles, df 1,29 F=3.05, p=0.005) (Fig 4.21). Burrows were non-significant but quadratically related to number of water vole captures (Regression equation: Burrow= 4.103 – 1.037water voles + 0.4707water voles^2, df 2,29 F=3.31, p=0.052). However, the low T/F and R^2 values for each field sign (latrine: T=2.9, R^2=23%; feeding station: T=3.1, R^2=25%; burrow: F=3.31, R^2=20%) indicate that only a small portion of the variability is explained by the regression line but this could be down to small sample size or a caveat of sampling a linear sampling in a non-linear site.

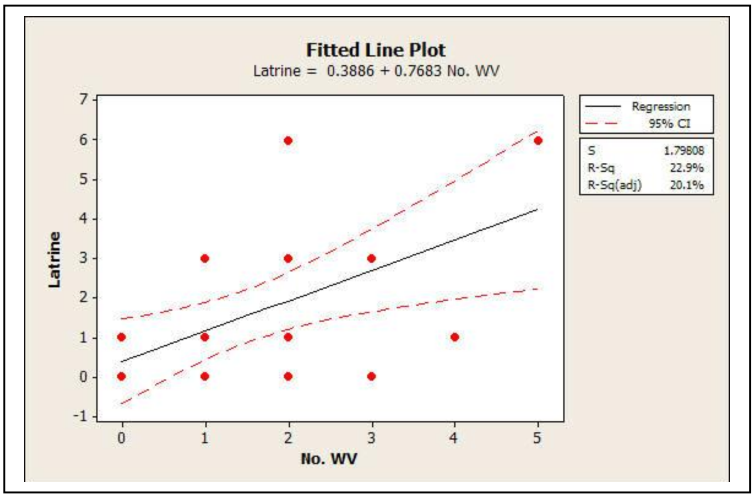

Fig 4.20: Regression analysis of latrine abundance as a predictor of water vole numbers

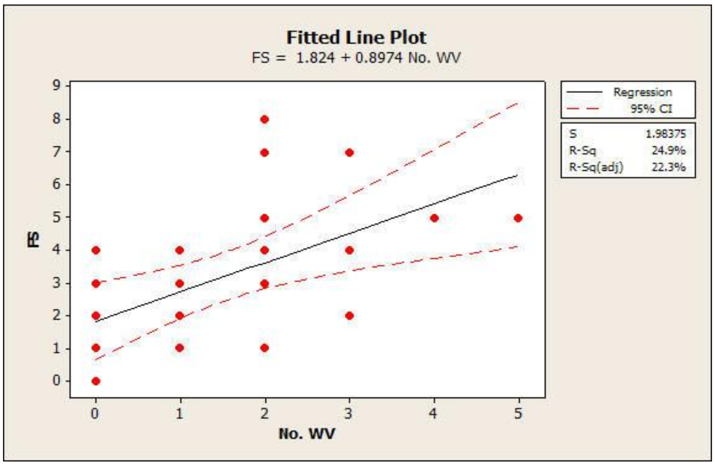

Fig 4.21: Regression analysis of feeding station abundance as a predictor of water vole numbers

4.3 - Habitat Classification and Plant Communities: Trapping Sites

4.3a Phase 1 Habitat Classification

All trapping sites were classed as B2 (neutral grassland) under the Phase 1 Habitat Classification (Joint Nature Conservation Committee, 2014). All sites were urban and have been created, or at the very least, significantly impacted by human modification. All four have undergone some type of human-induced alteration to sward composition to varying degrees of intensity (e.g. addition of herbicides, grass cutting regime, etc). Cranhill Park has received the greatest amount of agricultural improvement historically compared to Avenue End, Garthamlock or Tillycairn.

All trapping sites were dominated by grass species and low in plant species diversity (see Table 4.10). *Holcus* were the dominant grass species at Garthamlock, Cranhill and Tillycairn with a mean percentage cover of over 45% at each site. Avenue End was dominated by *F. rubra* (28% cover) followed by *H. lanatus* (12%). Average sward height was 30cm for Avenue End, 35cm for Garthamlock, 35cm for Cranhill Park and 45cm for Tillycairn (see Appendix IV).

4.3b National Vegetation Classification

Avenue End was classed as MG9, ill-drained *Holcus-Descampsia cespitosa* grassland with some water-logging. Garthamlock and Tillycairn were classed as MG1 *Arrhenatherum elatius-Festuca rubra* sub-community, a very species poor community dominated by tall, tussock grasses. Due to the alteration of sward composition, Cranhill could not be classified by the NVC system as species-poor grasslands dominated by *H.lanatus* and *H. mollis* do not fit into the current system (Averis, 2013).

Table 4.10: Plant species and mean percentage cover recorded in 2 x 2m quadrats at trapping sites

Plant Species	**Avenue End Primary (% cover)**	**Garthamlock Water Towers (% cover)**	**Cranhill Park (% cover)**	**Tillycairn Drive (% cover)**
H. lanatus	12	48	57	13
H. mollis	-	-	3	32
D. glomerata	-	-	-	12
F. rubra	28	4	-	10
Agrostis	11	10	35	5
A. elatius	-	5	-	7
D. cespitosa	3	-	-	-
Cynosurus cristatus	5	-	-	-
Cirsium arvense	-	13	-	8
Ranunculus acris	2	1	-	4
Ranunculus repens	9	7	1	1
Epilobium montanum	1	-	1	3
J. effusus	1	-	-	-
Equisetum arvense	3	5	-	-
Centaurea nigra	2	-	-	-
Carex sp.	3	-	-	-
Rumex acetosa	1	-	-	-
Trifolium repens	2	-	-	-
Lathyrus pratensis	1	1	-	-
Moss	10	-	-	-
Total Species	**16**	**8**	**5**	**10**

5. DISCUSSION

5.1 Riparian or Fossorial Populations?

Although fossorial populations are a common occurrence in Europe, they are rare in Britain. Lawton and Woodroffe (1991) stated that breeding water vole colonies could "never" be supported by habitat away from water. Other studies have shown that water voles rarely move more than 1-2m from the river bank (Stoddart, 1970; Lawton & Woodroffe, 1991) and habitat management guidelines for the species state that for a site to be suitable water must be present all year round (Strachan, 2004). The critical question of this study was: have the East End water vole populations adapted to living in grassland habitat and can they be considered an ecologically distinct fossorial population? To answer this fully several key aspects must first be examined: the distance from water, the level of water vole habitat use and any barriers to movement between patches of adjacent riparian and grassland habitat.

Results of this study show that 43% of sites occupied by water voles in Spring and 38% of sites in Autumn were within 0-150m of suitable riparian habitat. Sites at > 550m accounted for 33% of occupied sites and they remained occupied between Spring and Autumn surveys. When field signs were subdivided by the level of water vole habitat use, eight of the nine sites within 150m of riparian habitat showed feeding behaviour, with the ninth site showing historical water vole presence. A total of 9 sites supported breeding colonies, six of which were >550m from riparian habitat, the exceptions being Hogganfield Park grassland (adjacent to reed beds), the Fort Green Wall (336m from Auchinlea pond which was negative for water vole presence) and one site on the M8 corridor embankment.

This leads to the question: at what distance from riparian habitat can a water vole be classed as fossorial? If grasslands are adjacent to riparian habitat and a water vole can move freely between both is it possible to even assign a specific ecotype? The possible reasons put forward as to why East End populations have moved into grasslands were as follows:

1. Grasslands are unfavourable and only used opportunistically during times of high population density by the semi-aquatic ecotype or by juveniles forced to use grassland to disperse.
2. East End populations are expressing ancestral fossorial behaviour which originates from their founding European relatives and are capable of using both riparian and grassland habitats.

Evidence from this study points towards the occurrence of two ecologically distinct populations of water voles in the East End: one in close proximity to riparian habitat which uses grassland opportunistically and the other which exists and breeds in grasslands independent of riparian habitat. This was strongly supported by habitat preference modelling which revealed water vole distribution did not decrease with increasing distance away from riparian habitat as would be expected for a semi-aquatic animal: the likelihood of occurrence was equal at sites within 0-150m of riparian habitat and sites over 550m distant but reduced at the intermediate distance of 152-550m. In addition to this, field sign evidence indicated that the populations within 150m of riparian habitat were using grasslands for feeding only. Given their close proximity to riparian habitat and level of habitat use it is possible to conclude these water voles are the semi-aquatic ecotype but are capable of using grassland habitat opportunistically.

In contrast, six out of the nine East End breeding sites were at a minimum distance of 550m from water, five of which were over 700m. However, eight of these sites including the Fort Green Wall, are considered to be fossorial populations given their distance from riparian habitat and the highly fragmented surrounding environment which would restrict movement and dispersal. The presence of latrines on the nine sites unequivocally proves the presence of breeding colonies and not transient individuals. Latrines have long been considered the definitive field sign for breeding because of their association with territoriality (Woodroffe *et al*. 1990b; Strachan & Moorhouse, 2006). The eight fossorial breeding colonies were within the city of Glasgow, associated with the M8 corridor and concentrated along a 3km stretch of the road embankment and adjacent patches of grassland. The Fort Green Wall was less than 550m distant from riparian habitat but water voles were still considered to be fossorial here because the site is severely restrictive to movement, being sandwiched between the shopping centre to the north, motorway to the south, car park to the east and broadleaf woodland to the west.

The ranges of movement quoted by Stoddart (1970) and Lawton and Woodroffe (1991) arise because water vole distribution is thought to be limited by the availability of riparian vegetation. Water voles are known to feed on numerous grass species though (Strachan & Moorhouse, 2006; WildCRU, 2002) and the sites between 0-150m of riparian habitat may have been used to maximise foraging opportunities. The area of grassland habitat available to water voles was greater than the area of riparian habitat, which was concentrated in a strip surrounding the water body, therefore, if population densities were high it could force individuals to expand their range should food become a limiting resource. Radio-tracking studies have shown water voles are highly active within their home range and capable of moving distances far greater than previously thought, over 100m in less than an hour (Moorhouse & MacDonald, 2005). The fact that water voles are infrequently recorded at any distance from the riparian margin in previous studies could be more a result of unsuitable neighbouring habitat than a dependence on water e.g. water voles are known to actively avoid riverbanks adjacent to intensively managed farmland (Strachan & Jefferies, 1993).

The frequency of occurrence of feeding stations in Autumn was double that of Spring surveys, which may have coincided with an increase in the population after the breeding season. It is possible that the field signs were caused by dispersing juveniles moving through grassland as they migrated in search of unoccupied riparian habitat. Radio-tracking of 12 individuals by Moorhouse & MacDonald (2005) found they dispersed distances of 186-949m. Telfer *et al*. (2001) showed that water voles can disperse across land and found field signs of fossorial animals in bracken, meadow and heather (Telfer *et al*. 2003a). Dispersal documented by Fischer *et al*. (2009) in lowland metapopulations showed a "stepping-stone" pattern where radio-tracked juveniles were found to stop temporarily in a habitat patch, for up to one week, before continuing on. The daily movements varied considerably between individuals, ranging from 18m to 1800m (Fischer *et al*. 2009). However, a study which calibrated field sign indices with water vole abundance found that at low population densities, only feeding field signs were recorded (Woodroffe *et al*. 1990) so it is possible that the field signs found were from a small number of fossorial water voles and not the opportunistic or dispersing riparian ecotype. Without investigating spatial movements between habitat patches by CMR or radio-tracking, the exact ecotype of the water voles on sites within 150m of riparian habitat remains ambiguous.

The urban nature of the East End creates a mosaic of habitat patches which are highly fragmented with significant barriers to movement such as housing estates, roads, and heavily managed grassland areas. Although fragmented lowland metapopulations can be spread over 550-980m (Telfer *et al*. 2001) the area occupied by an actual breeding colony is considerably smaller and individual territories smaller still. Even males with their larger home ranges only tend to occupy a maximum length of 300m (Strachan & Moorhouse, 2006). With distances of over 550m, it seems highly improbable that water voles could be capable of moving freely between riparian and grassland patches, particularly given the complexity of the non-linear habitat matrix in the East End or have a home range spanning such a distance. Giraudoux *et al*. (1995) did note that fossorial water voles were sometimes forced into unsuitable habitat when numbers peaked during population cycles. However, a habitat must be of sufficient quality in order for breeding to be successful (Lawton & Woodroffe, 1991) and classing grassland as unsuitable would be highly unlikely taking into account water vole densities encountered at trapping sites.

Transitional forms of the water vole which are capable of using both riparian and grassland habitat are common in Europe and individuals can display both the fossorial and riparian ecotype within their lifetime. The fossorial ecotype occurs in upland meadows in mountainous regions of the Pyrenees and Alps and the transitional form occurs south of the Alps (Wust-Saucy, unpublished; Taberlet *et al*. 1998). Genotyping by Piertney *et al*. (2005) revealed that Scottish populations are descended from French, Spanish and Swiss colonisers, therefore, it is possible that the populations which colonised Scotland during the last Ice Age were from this transitional genetic clade and the fossorial lifestyle of the East End water vole is the expression of an ancestral behaviour. Other fossorial Scottish populations have been recorded (Eilean Gamhna -Strachan & Moorhouse, 2006; Sound of Jura - Telfer *et al*.

2003a) and it may be the case that for fossorial behaviour to manifest certain biotic and abiotic conditions must first be in place, similar to the conditions which promote fossorial populations in Europe i.e. high density riparian water vole populations and readily available suitable grassland habitat.

Grasses were the dominant vegetation, comprising over 70% of plant species, on sites with breeding colonies in Lawton & Woodroffe's study (1991) and grasses and sedges were equally likely to be the dominant vegetation type in upland rivers in the Cairngorms (Capreolus Wildlife Consultancy, 2005). Aars *et al*. (2001) also found an association between grasses and water vole presence. The ready usage of grass-dominated habitat by water voles is evidently well documented in Britain and the occurrence of fossorial behaviour could very well be commonplace. Indeed, the fossorial and/or transitional ecotype may be massively under-recorded in the UK because standard methodology restricts surveying to wetland habitat types, making the Jura and East End populations unusual but probably by no means unique.

5.2 Water Vole Distribution

A total of 32% of sites surveyed were occupied by water voles in Spring and 31% of sites occupied in Autumn 2014. Of these, almost half were considered to be occupied by riparian populations and a third by fossorial populations. This indicates that the East End of Glasgow supports both and should be considered a highly important stronghold for water voles in light of their overall sparse distribution in the Scottish Lowlands and protected status. However, it is important to note that without investigating water vole distribution and population density between neighbouring riparian and grassland patches the relative importance of grassland compared to riparian habitat is impossible to quantify.

The M8 motorway is clearly an important landscape feature in relation to the distribution of fossorial populations. The eight fossorial breeding colonies were all associated with the M8: four were spread linearly along the M8 embankment, another two on adjacent derelict sites, one in Cranhill Park, and one in a children's play area in Easterhouse. Given 62% of all occupied sites were within 1km of the M8 corridor and that distribution was linearly related to distance from the motorway it is possible that the populations on these sites function as a metapopulation. Telfer *et al*. (2001) proved that it was possible for lowland metapopulations to function and persist when separated by 550-980m. The relationship between the M8 and water vole distribution also gives weight to the theory that the fossorial behaviour displayed by the East End water voles originated when populations along the Old Monkland Canal were forced into surrounding grasslands as a result of construction work on the M8 in the early 1960's. However, without mapping the distribution of populations to the south of the M8 corridor it is difficult to speculate and the origin of these populations would be better answered using genetic methods.

There is a strong probability that fossorial breeding colonies are numerous throughout the East End albeit concentrated along the southern edge of the study area. Tillycairn was not included in the presence/absence surveys but trapping attested to the presence of breeding water voles. Lawton & Woodroffe (1991) found colonies tended to exhibit clumping: if a colony was already present in a habitat, it increased the likelihood of neighbouring colonies. The stratified surveying methodology used in this study was adopted to cover landscape scale water vole distribution and encompassed $34km^2$ but invariably resulted in the actual number of water vole colonies within each $1km^2$ PSU grid square being under-represented. Transect distances of 100m would also result in smaller, potentially occupied patches being missed. Despite these limitations, the stratified approach was both a time and cost effective method for covering the study area and the distribution maps are considered a highly accurate representation of the species range throughout the study area given that each PSU was systematically walked on foot. False positives rarely occur when conducting presence/absence surveys but some degree of under-recording is thought likely particularly if population densities are low or where landscape features make sampling more difficult (Gu & Swihart, 2003). Although Autumn surveys showed a decrease in the number of sites from 21 to 19 this could be attributed to non-detection rather than sites no longer being occupied. It is possible that field signs were missed on these two sites (located in Robroyston Park) because they were dominated by dense grass species and weeds.

5.3.1 Grassland Habitat

Woodroffe *et al*. (1990) hierarchically ranked water vole habitat value based on the intensity of habitat use and summarised that breeding colonies must select optimal habitat in order to sustain population levels. However, habitat selection and preference are nowadays considered to be far more complex (Beyer *et al*. 2010). The relevance of the intensity of habitat use is largely based on several assumptions: that habitat value is directly related to the level of habitat use and that all habitat types are equally accessible and available throughout the environment (Berg, 2004). Whilst it may be difficult to evaluate the quality of grassland habitat for water vole populations precisely, the high abundance of breeding field signs and number of animals captured during trapping infer that grasslands are valuable habitat for water voles. The population density estimates calculated from CMR and population modelling in this study were simplistic because of the linear trapping methodology used but never the less still indicate that the East End grasslands may support some of the highest water vole densities ever recorded in the UK.

Population density is directly influenced by food quality (Cockburn & Lidicker, 1983; Moorhouse & MacDonald, 2008) which demonstrates that the plant species in the grasslands of the East End are palatable to water voles and nutritious enough to sustain high densities. Only 48 water voles were trapped during this study but the animals encountered were all in good body condition, except for two individuals, which adds further support to grasslands being a valuable habitat for the species. Mate *et al*. (2013) also found that vegetation quantity, quality and composition were critical to the reproductive success of the Southern water vole *A. sapidus*. As well as influencing reproductive success, better food quality can result in earlier breeding attempts (Taitt & Krebs, 1983; Moorhouse & MacDonald, 2008). Whilst only a stand-alone incident, the dissection of a dead female killed by machinery in Garthamlock on the 3rd March 2014 revealed she was pregnant with three offspring. Water voles with a more northerly range are thought to start breeding at the end of April (Strachan & Moorhouse, 2006) but it is possible that urban grasslands provide high enough quality food to enable earlier breeding in East End populations.

5.3.2 Water Vole Habitat Preference

The distribution of East End water vole populations was strongly associated with urbanised habitats and along with distance from water, habitat type was the only explanatory variable revealed by habitat modelling. Water vole habitat preference was for parks and associated grasslands (e.g. golf course, garden) which were all heavily modified for human use and subject to moderate to high levels of habitat management, whereas, rank grassland and similar habitat types (e.g. field verge, agricultural land) appeared to be actively avoided. Parks tended to be lower in botanical diversity compared to the other habitat types but the intensity of grassland management promoted the dominance of *H. lanatus*, a grass species considered to be fast growing and disturbance-tolerant (Rodwell, 1992). Given the correlation between occupation and the occurrence of *Holcus* species, it is evident that water voles displayed a strong association with these particular grasses. Habitat preference could in part be caused by the high incidence of *Holcus* species as the dominant grass species in park habitat type. Numerous studies show water voles require dense vegetation to provide cover and food (Lawton & Woodroffe, 1991; Macpherson & Bright, 2011). Both *H. lanatus* and *H. mollis* are dense, tussock-forming grasses which are highly palatable to grazing animals (Averis, 2013). At water vole hotspots such as Cranhill, the grasses formed thick mats carpeting the area which appeared to be a characteristic specific to *Holcus* species and highly favourable to water voles.

Parks also tended to fall into the moderate to high disturbance class given their use for recreational activities by people (e.g. dog walking, football). Small mammals are considered to be “urban-adapters” and alter their activity patterns in order to cope with human disturbance (McKinney, 2002). It is possible that the fossorial behaviour displayed by the East End populations has arisen in part due to their proximity to humans and apparent disturbance-tolerance. Sites with moderate to high disturbance were less likely to record fox field signs and it is possible that water vole populations indirectly benefit from lower predation because of their occupation of disturbed habitats. Fragmented water vole populations are highly vulnerable to predation (Rushton *et al.* 2000) and the frequency and abundance of fossorial populations in the

East End may be partially explained by the combined effect of disturbance-tolerance and low predation rates.

Rank grasslands were all subject to low or non-existent management and tended to be associated with more semi-natural or natural environments. They were mainly classed as low disturbance with the exception of vacant land which was classed as moderate disturbance. Botanical diversity was varied at these sites and the lack of management meant that weeds and waterlogged soil were common features. The dominant grass species of rank grasslands were predominantly *D. cespitosa* and *Agrostis* species (52% of all unoccupied sites). *Agrostis* species were the third most frequently recorded species on occupied sites which at first glance appears contradictory, however, *Agrostis* species are fast-growing colonisers and were co-dominant on a number of sites. While they are palatable to water voles (Strachan & Moorhouse, 2006) their shorter height and patchier growth does not provide the dense cover required by water voles. *D. cespitosa* is tussock-forming and has been recorded as a water vole forage species (WildCRU, 2002) but in this study its occurrence appeared to be negatively related to water vole occurrence. This could be down to two possible factors: *D. cespitosa* has a broad tolerance of environment conditions (Averis, 2013) and may have be more frequently recorded on north-facing sites because of its greater tolerance of colder, damper conditions. The leaves also develop a high silica content with age (Averis, 2013) which would also make them unpalatable to water voles.

Road verges, similar to parks, were associated with the urban environment but management levels are considered to be lower (e.g. less frequent grass cutting regime). They were always classed as moderate disturbance due to traffic. It is possible that road verges act as a refuge for water vole populations particularly in light of the following: the linear relationship between the M8 and occupied sites, the association of breeding colonies with the M8 and the high percentage of road verges as neighbouring land use next to occupied sites. The importance of road verges as a wildlife corridor has long been recognised (Bennett *et al*. 2006; Ascensao *et al*. 2012) and they can function as a refuge for numerous small mammal species (Adams & Geis, 1983; Bellamy *et al.* 2000) as well as bees (Hopwood, 2008), butterflies (Saarinen *et al*. 2005) and birds (Meunier *et al*. 2000).

As previously mentioned though, habitat preference is complex and a degree of caution is necessary when assuming preference based on solely on occupation. In the East End, core colonies were concentrated at the southern-most point of the study area (along the M8 corridor) within the council district of GCC and there was a clear divide in the level of urbanisation between GCC and North Lanarkshire district to the north. Habitat types in North Lanarkshire were predominately rank grassland or agricultural; gardens, parks and derelict sites were only recorded within the boundary of GCC. Therefore, not all habitat types were equally available and non-occupation of northern sites may instead be a confounding factor of isolation rather than an active water vole preference for a particular habitat type. The fact these urban sites support water vole populations is unequivocal though.

Given the high degree of habitat fragmentation in the East End, the water vole populations here will most likely function as a metapopulation and display the same demographic trends of extinction and re-colonisation as other metapopulations (Aars *et al*. 2001; Telfer *et al*. 2001). The most important variable determining occupancy in metapopulations, in the absence of mink, is not habitat type but isolation, with the chances of occupancy decreasing with increased distance away from breeding colonies (Lawton & Woodroffe, 1991). Population modelling by Telfer *et al*. (2001) also found isolation and habitat quality to influence the likelihood of site occupation.

Isolation distances between occupied and non-occupied sites measured 1.2-4.2km (measured as direct over-land distances) and barriers due to the habitat matrix would impede dispersal, limiting opportunities for the water vole population to expand northwards. For overland isolation distances, Telfer *et al*. (2001) found re-colonisation did not occur at distances greater than 1.1km. The isolation distances necessary for metapopulation persistence of upland riparian colonies was reported as 0.2-1.6km by Aars *et al*. (2001) and 0.4-0.7km by Capreolus Wildlife Consultancy (2005). The breeding colonies along the M8 were separated by 0.1-0.8km. No study has been done on metapopulation structure or persistence time in urban water vole populations and further research is required to understand the population dynamics of East End water voles.

The model used for water vole habitat preference in this study was simplistic and there is a high likelihood that some important explanatory variables were missing. Binary logistic regression is considered a robust statistical method for habitat preference analysis (Keating & Cherry, 2004) but using it in this instance resulted in a number of environmental variables being excluded from the model because of categorical variable counts falling below five. There is a strong likelihood that the excluded finer-scale environmental variables (e.g. slope profile) may also factor in water vole habitat preference. For example, preference for a steep bank profile has previously been documented in riparian water voles - a behaviour which is thought to avoid the risk of burrows flooding when rivers are in spate. Lawton & Woodroffe (1991) found breeding colonies displayed a preference for a slope of 35º and Capreolus Wildlife

Consultancy (2005) finding a preference for slopes of 17º although Lambin *et al*. (1998) found no correlation between water vole presence and slope profile. Only 12% of all sites surveyed in this study had a slope profile greater than 16º but 50% of these were occupied and flat sites consistently showed low occupancy (20% in Spring surveys; 22% in Autumn surveys) which indicates that East End populations may well prefer steeper slopes and warrants further investigation. The fact that some of the East End populations are fossorial and not at any risk of flooding means this preference would be of little adaptive benefit, however, it may still persist because steep banks could be correlated with other favourable habitat criteria such as vegetation height or bank penetrability.

In this study 63% of occupied sites had a mix of short and tall grass species (translating as grassland dominated by tussock-forming grass species with a structurally complex sward) but grass layering had to be excluded from the model because of counts below 5. Lawton & Woodroffe (1991) measured layering within vegetation structure and found a strong preference for mid to high layering. Both Telfer *et al*. (2001) and Capreolus Wildlife Consultancy (2005) reported total cover as more important than layering. A structurally complex sward is highly likely to be an important factor in water vole habitat preference and should be further investigated in the East End populations.

While not an explanatory variable for habitat preference, neighbouring land use may still be important for water vole distribution as it could be indicative of movement or dispersal pathways, or lack of, between occupied patches e.g. a site with adjacent parkland has far greater potential for water vole dispersal compared to a site with an adjacent housing estate. Indeed, water vole field signs were noted to follow a stepping-stone pattern between sites occupied by breeding colonies and surrounding patches of grassland (Robyn Stewart, *pers. ob.).*

5.4 Population Densities and Trapping Methods

Findings from this study indicate that the urban grasslands of Glasgow's East End may sustain some of the highest densities of water voles ever recorded in the UK. While the estimates are fairly crude because of the limitations of the linear trapping methodology used, Cranhill Park (which was dominated by highly favoured *Holcus* grass species) appeared to be a water vole hot spot with the highest density estimates of 156 individuals per ha. In contrast, previous UK studies estimated reed bed population densities at 40-50 per ha (Strachan & Moorhouse, 2006) and the fossorial populations of Jura had an average density of 26 water voles per ha although this did increase to 70 individuals in Spring (Telfer *et al*. 2003). These densities are still small relative to the peak European population densities of 1000 per ha (Meylan, 1977; Giraudoux *et al*. 1995). However, some caution seems necessary in interpreting the East End population estimates due to the variation between the two trapping sessions at Cranhill. The August population density based on modelling estimated numbers at 84 water voles per hectare which is half the July estimate of 156 water voles. This decrease in density may be an indication of a decline in the population or may perhaps be accounted for by behavioural changes or sampling artefact.

Fragmented water vole populations are particularly vulnerable to extinction from both environmental and demographic stochasticity, especially predation (Rushton *et al*. 2000). Although the July trapping was cut short by human interference, water voles still occupied the site and there was no evidence of fatalities, nor any signs of major environmental change (e.g. fire, flood), predation or dug out burrows when trapping re-commenced four weeks later. Weather from May through to August 2014 was characterised by stable high pressure systems, above average temperatures and normal seasonal rainfall (MetOffice, 2015). Large sections of desiccated grass with little fresh growth were observed at the trapping site in August (Robyn Stewart, *pers. obs.).* It is possible that a number of water voles emigrated from the site between trapping occasions in search of lusher grasses to feed on or allocated a greater proportion of their foraging time to eating roots and rhizomes along underground runs meaning they were less likely to be trapped. Cranhill Park is managed for water voles by GCC with four patches of long grass within the park. It was unknown how freely water voles move between these habitat patches. While only an anecdotal observation, the August session was plagued by slugs given spells of overnight rain when the trapping took place. The bait used appeared to be attractive to slugs and while it is impossible to quantify, a number of water voles may have been deterred from entering the traps because of the high numbers of slugs.

Behavioural changes in the water voles could account for the decrease in the population estimate. Disturbance caused by the July trapping could have resulted in a switch to more subterranean activity patterns. This behaviour could be common to all grassland populations across the East End and it is possible that the same decline in numbers would have been mirrored at Tillycairn if a second trapping session had been carried out. The water vole populations of Cranhill are continually exposed to high levels of disturbance by walkers, dogs and other park users and disturbance was witnessed on almost every occasion the Park was visited. Disturbance-tolerance or disturbance-avoidance behavioural mechanisms such as less time above ground would be an important adaptation for populations in the East End to develop, particularly in response to random, short-term disturbance events. The likelihood of such disturbance-tolerant behaviour is further supported by habitat preference modelling which revealed a strong water vole preference for parkland which was characterised by moderate to high disturbance levels.

Another possibility could be that the July trapping represents a peak in breeding behaviour when the water voles were exceptionally active. Females are territorial during the breeding season (Strachan & Moorhouse, 2006) which may have resulted in higher trapping likelihood while they were patrolling and maintaining latrines. Females can have a succession of litters during the breeding season and the quantity of time spent above ground will be dictated by whether they are nursing young or seeking a mate. The August trapping could have coincided with multiple females nursing young making them less likely to be active above ground (Strachan & Moorhouse, 2006). However, the total number of females trapped were 15 and 11 respectively, so this seems unlikely. Males accounted for the greatest drop in numbers, falling from 10 to 2 individuals, which may indicate a strong behavioural difference between the sexes with males being highly sensitive to disturbance and becoming trap-shy after July. Only one male was recaptured for the entire study (at Tillycairn) and females were significantly more likely to be trapped during the day overall compared to males which provides further support for a sex-based difference in trapping likelihood. However, with a sample size of only 48 further research is required to fully investigate this before any conclusions can be drawn.

Recapture rates at Cranhill were low, 16% in July and 15% in August, meaning only a small percentage of the total population were marked and such sparse capture histories may have resulted in less robust population estimates (Amstrup *et al*. 2005). Recapture rates at Tillycairn were 60% for adult water voles (juveniles were considered non-resident within the habitat patch because of the likelihood of dispersal and excluded from analysis) and the low recapture rate does appear to be site-specific to Cranhill. Aars *et al*. (2006) found recapture rates of 73-92% over a 4 day trapping period in upland Scottish populations. European fossorial water voles are quoted as being very easily trapped with up to 70% of the population captured in the first day but methodology involved deliberately disturbing burrow entrances to elicit investigatory behaviour from the animal which increased ease of capture (Meylan, 1977). Telfer *et al*. (2003a) found 3 days of trapping was adequate for the Jura fossorial voles but traps were set on fixed grids rather than the linear transect used in this study.

The accumulation curve for the number of captures also supports this with no clear sign an asymptote had been reached on any of the trapping occasions. Tillycairn and the August trapping at Cranhill both showed a gradual increase in the number of captures, indicating that a relatively high proportion of the water vole population in the sampling area was captured over the 5 days, but the July trapping at Cranhill proved anomalous again, with numbers doubling within a 24 hour period. The number of trapping hours was unequal between sites: 960 hours at Tillycairn, 728 hours at Cranhill in July and 600 hours in August. The difference of 128 hours between trapping occasions at Cranhill would invariably have reduced the potential for captures in August.

Methodology was replicated between all sites and trapping occasions with trap positions, etc selected by the same individual; human error is an unlikely explanation for the difference in recapture rates. PIT tags are a proven effective method of individually marking animals and have a high retention rate (Harper & Batzli, 1996;

Melis *et al.* 2011), therefore, PIT tag failure or loss is considered unlikely especially when factoring in the extensive experience of the marker. A trapping duration of 5 days may be adequate for the majority of rodent populations (Gurnell & Flowerdew, 2006) but not the fossorial water voles of the East End. It is recommended that any future work should increase the trapping duration but that the inherent risks of biological sampling in an urban environment should also be taken into account. This study was disrupted on multiple occasions: traps were interfered with, stolen, verbal threats were received while conducting field work and incidents of wildlife crime were witnessed.

Linear trapping along a transect line is a well established sampling technique for many small mammal species (Gurnell & Flowerdew, 2006) and is standard methodology for riparian populations of water voles (Strachan *et al*. 2011). While it proved an effective method for initial research, sampling on a grid square pattern similar to Telfer *et al.* (2003a) would be recommended for any future work. Linear sampling was adopted because of cost constraints and concerns over sampling in the urban environment. Any trapping attempts prior to this study have been unsuccessful. Sampling on a grid pattern would allow for the collection of information on individual spatial movements and provide insight into home range and territory size. A large number of traps would be required to sample on any meaningful scale. All recaptures occurred at the same point on the trapping transect but this is most likely an artefact of linear sampling in a non-linear habitat rather than an indication of home range size. Stoddart (1970) found that a trapping grid of 15 x 20m was not large enough to encompass the entirety of a riparian water vole's range but scans of the subterranean burrow system using ground-penetrating radar at Cranhill revealed that a complete burrow system occupied an area of 8 x 10m^2 (Stewart *et al*. in preparation for publication). Whilst this was only part of a pilot study and further work is required, it does indicate that grassland water vole territory size can be small at high population densities. The mean range size of water voles has been shown to decrease in response to higher quality foraging (Moorhouse & MacDonald, 2008) so it may be possible that considerably smaller home range sizes can supported in grasslands compared to riparian habitats.

It is difficult to draw any robust conclusions from trapping regarding water vole morphology or behaviour given the small sample of 48 and non-uniformity between trapping occasions. The water voles trapped during this study had a mean body mass of 108.5g which is below the normal UK range of 140-350g (Strachan *et al*. 2011) and may point towards smaller body size in grassland populations. A number of large males were captured however (maximum body mass 221.7g) and a wide range was evident in the biometric measurements. European fossorial water voles tend to be smaller in size but the range of 60-150g quoted by Saucy (1994) could actually be for *A. scherman* rather than *A. amphibius* because it pre-dates Panteleyev's (2001) separation of the species based on morphological adaptations. The biometrics of fossorial East End water voles should be directly compared to neighbouring riparian populations to investigate this fully. Juveniles were only trapped at Tillycairn which is most likely due to the timing of trapping (28th July to 1st August) coinciding with the time of dispersal of the newly emerged juveniles.

Riparian water vole diel pattern show both activity during darkness and daylight hours but the majority of captures are thought to occur predominately overnight (Strachan & Moorhouse, 2006). The results of this study point towards more diurnal activity patterns and a higher trapability of females but again, no meaningful conclusions can be drawn for the East End populations because of the small sample size and non-uniform trapping. Trapping probability was higher between 0530 and 1300 compared to overnight captures but this was only found during August trapping at Cranhill; Tillycairn was non-significant. Being more active during the day may mean that water voles are less likely to be predated on if their main predators are nocturnal e.g. fox. Camera trap footage also indicated a higher level of diurnal over nocturnal activity.

5.5 Predation and the Presence of American Mink

Water vole populations in the East End were found to have numerous predators: domestic cat, fox, domestic dog (Robyn Stewart, *pers. ob*.) and carrion crow (*Corvus corone*) (Robyn Stewart, *pers. ob.*). Foxes were the most frequently encountered predator, on a third of all sites, and it is highly likely foxes are the main predator of the East End populations. European fossorial water voles can make up 54% of the items in a fox's diet (Weber & Aubry, 1993). At Garthamlock Water Towers fox scat was extensive across the site (although scat were not analysed) and approximately 30% of water vole burrows appeared to have been dug out, indicating that fox predation may have a considerable impact on local populations.

The impact of generalist predators like fox, stoat (*Mustela erminea*), badger (*Meles meles*) and the long-eared owl (*Asio otus*) on water vole populations is well documented (Weber *et al*. 2002) but there is little existing literature on the impact of domestic pets on urban populations. Weber *et al*. (2002) considered domestic cats to be a significant predator of European fossorial populations. Domestic cat numbers are estimated at 8.5 million in the U.K (RSPCA, 2014) and while greater concern has been expressed over their predation levels of bird populations, a questionnaire conducted by The Mammal Society in 1997 of 618 cat-owning households showed 69% of food items brought home were mammals, although water voles were a small percentage of the total (Woods *et al*. 1997). There are reliable eyewitness accounts of both cat predation at the Avenue End site (Geraldine O'Donnell, *pers. comm*.) and dog predation (Robyn Stewart, *pers. ob.*) and it is likely that domestic pets are responsible for a number of water vole fatalities.

Mink presence was negative for all sites on both survey occasions, however, there is no evidence that water voles and mink have ever co-existed in the area. Carter and Bright (2003) showed non-linear habitat such as reed beds could act as a refuge for water vole populations as mink rely on water courses for hunting. It was reasoned that the East End populations might have moved into grassland from neighbouring riparian habitat as an anti-predator adaption in response to the presence of American mink with grasslands acting as a refuge habitat similar to reed beds.

As the East End water voles have not been subjected to mink predation no causal relationship can be concluded because non-detection does not equate to absence. Survey methodology required all sites to be a minimum distance of 50m from riparian habitat in order to reduce the likelihood of encountering the riparian water vole populations. As an indirect result of this mink presence may have been missed because only grassland patches were surveyed. Adjacent riparian habitat should have been surveyed for definitive proof of mink presence.

Radio-tracking studies show mink forage within 10m of water courses (Macpherson & Bright, 2010) which indicates non-detection was a strong likelihood, however, records dating back to 1962 (GNHS, unpublished) show that historically mink have never been recorded in the East End. The closest sighting recorded was over 6km away although mink can disperse distances of 20-40km and do not have to follow water courses (Strachan *et al*. 2001). Mink are capable of devastating water vole populations within a very short time period (Telfer *et al*. 2001) and a sighting was reported at Johnston Loch in Sept 2014. Although this was unverified it could indicate that mink have spread into the Seven Lochs Wetland Park from the east, perhaps following the Bothlin Burn. However, records of fossorial populations date back to 2008 which indicates living in grasslands pre-dates the unverified mink sighting.

5.6 Seasonality of Field Signs and Field Sign Indices

Water vole field signs are highly distinctive and presence/absence surveys based on field signs have proved to be an effective method for establishing water vole distribution and abundance in both fossorial populations (Telfer *et al.* 2003a) and riparian populations (Woodroffe *et al*. 1990; Strachan & Moorhouse, 2006). The detection of different types of water vole field signs was significant between seasons with latrines and nesting material only present in Autumn surveys because of their association with breeding behaviour. The frequency of occurrence of both tumuli and feeding stations increased throughout the year which could represent either an increase in the population after the breeding season or that water voles change the proportion of time spent above ground throughout the year, with above ground activity responding to the plant growing season. Both Stoddart (1970) and Carter & Bright (2003) reported that water voles were largely subterranean during the winter.

Giraudoux *et al*. (1995) found a linear relationship between the number of tumuli and water vole abundance in European fossorial water voles but no relationship was evident in the East End population. A linear relationship was found between the number of latrines and water vole abundance which was in agreement with previous work, although estimates differed from those of Morris *et al*. (1998). Morris *et al*. (1998) regression analysis calculated 6 latrines per water vole whereas, this study found one latrine per water vole. Although both studies used linear sampling techniques, the East End water voles inhabit an area of grassland rather than a ribbon of riparian margin which could account for the difference. It may not be possible to directly compare the field sign indices between linear and non-linear populations. Invariably, counting the number of latrines within a 2.5m radius at each trapping point would give a lower count compared to a continuous survey along stretch of riverbank. The number of trapping sites is also likely to be a contributory factor giving a considerable margin of error due to small sample size. This was particularly evident in the widely scattered data points of the regression line in Fig 4.21. A number of underground latrines were uncovered during building work in Garthamlock which again points towards distinct fossorial behaviour in the grassland populations and could also account for the fewer incidences of latrines.

5.7 Management Implications

Establishing the distribution and habitat variables of water vole populations across the East End was the first step towards pro-active conservation management of this species yet there are still questions regarding their origin, genetic diversity, population dynamics, behaviour and life history traits. Population densities alone point towards the East End water voles being of national significance and they should be considered a key regional area for the future conservation of this species in the UK. No distinction should be made between riparian or fossorial populations because of the high probability of East End water voles switching between habitat types – the evolutionary heritage of both may be interlinked and of equal importance. Even with the current lack of genetic research on these populations, there is a strong argument for their consideration as an Evolutionary Significant Unit (ESU) on the basis of their population and ecological distinctiveness (Crandall *et al.* 2000). It is highly likely that gene flow between breeding colonies is restricted due to isolation distances and the urban habitat matrix and these populations may show genetic variation from other Scottish populations. However, this does not necessarily mean that East End populations will be effected by inbreeding and the period of time these populations have been isolated for remains unknown. Genetic diversity can be maintained within metapopulations (Stewart *et al*. 1998). Stewart *et al*. (1998) found the genetic composition of metapopulations fluctuated on an annual basis and gene flow was more effective within populations rather than between them. A note of caution for the future genetic viability of the East End populations should be taken from Telfer *et al.* (2003b) study though – only 50% of microsatellite polymorphism remained in these highly isolated island populations compared to mainland populations.

Understanding metapopulation dynamics and connectivity between habitat patches is probably the most pressing concern because of the severe dispersal barriers and isolation distances of the urban environment which makes these populations highly vulnerable to extinction. Bright (1993) warned of the long-term consequences of the unstoppable trend towards urbanisation on the distribution of small mammal species due to their shorter dispersal distances. Without investigating metapopulation dynamics between the East End populations there is no way of knowing their long-term viability. The 2014 surveys showed that overall there was no significant change in population distribution, however, no robust conclusion about population trend can realistically be drawn from a one year study.

The East End is undergoing a timely redevelopment programme co-ordinated by the Scottish Government and GCC to improve the standard of living in this socially deprived area. Land-use change is inevitable and will be extensive across the entire area. Water vole populations are at high risk of disturbance and being negatively impacted by the rate of environmental change which may be too rapid for normal metapopulation processes. Even the high population densities discovered at hotspots like Cranhill Park does not guarantee long-term survival of water voles in the area – space is a limiting factor here and dispersal distances may be insurmountable if connectivity between suitable habitats is not maintained.

The landscape approach is essential for conservation (Macpherson & Bright, 2011) because no one habitat exists in isolation and regardless of whether the East End populations should be considered an ESU has no bearing on the necessity for new management best practice specific to this population. Habitat creation, displacement, translocation and mitigation guidelines for development are currently based on the semi-aquatic ecotype (Strachan, 2004; Strachan *et al*. 2011). A water vole management strategy for the potentially transitional water vole populations should be a high priority for both the local authority and Scottish Natural Heritage and incorporate clear guidelines for developers, land owners and all other stake holders. Surveying techniques should be updated and disseminated between all relevant parties, the LBAP revised to include the findings of this study and an annual water vole monitoring programme established which incorporates both grassland and riparian habitats. The potential for human-animal conflict is considerable given the urban environment and the close proximity of water vole populations to people. Incidents of wildlife crime may be frequent and this urgently needs addressed e.g. dog-owners allowing their dogs to dig up burrows and kill water voles. Water voles have been reported in local gardens and community engagement to promote understanding of the presence and legislative protection of this species should also be a consideration.

Updating land management policies will be required particularly in relation to grass cutting regimes given the importance of dense cover for water voles, especially during the breeding season. Vegetative structure and plant composition change with the age of a grassland (Churchfield & Brown, 1987) and habitat management guidelines which maintain sward composition and includes known preferred grass species require development. Habitat creation schemes have proven successful if the specific requirements of that species are taken into account (Reid *et al*. 2007). These guidelines will be essential for informing future displacement and translocation programmes.

5.8 Conclusions

In conclusion, the grasslands of the East End of Glasgow appear to support two ecologically different populations of water vole: one within 150m of riparian habitat using grasslands for feeding and dispersal; the other existing independently of water and using grasslands for breeding. The latter population was considered to be ecologically different from riparian water voles given their distance from water, level of habitat use and the likelihood of restricted movement between riparian and grassland habitat patches due to the fragmented urban environment. These populations were classed as fossorial due to being located 500-1182m from riparian habitat and their non-transitory use of habitat i.e. their more mole-like existence and the successful establishment of breeding colonies. Exactly what caused the East End water vole populations to move into grasslands is still unknown but evidence from this study supports the theory that living in grasslands is the expression of an ancestral behaviour from their founding European colonisers. The populations here are most likely transitional water voles i.e. they can use both riparian and grassland habitat and cannot be defined as a distinct ecotype. It is possible that populations here are a relic of those once found along the Monkland Canal prior to it being filled in for the creation of the M8 which may in turn have forced water voles into surrounding grasslands as urbanisation increased. Their exact origin will remain a mystery until future genetic research is done.

The grasslands of the East End are largely a product of urbanisation; a fragmented, heavily modified environment created in the 1960's when suburbs such as Easterhouse and major infrastructure like the M8 were developed. Grassland water voles were almost exclusively associated with these urban habitats, living in parks, gardens, road verges and derelict sites. All occupied grassland patches were subject to moderate to high levels of management indicating that East End populations have adapted to the urban environment and developed some degree of disturbance-tolerance. Habitat preference modelling revealed that water voles preferred parkland habitat types and actively avoided rank grassland which is thought to be largely down to the management practices employed in parks which promoted the dominance of tussock-forming, palatable grass species like *H. lanatus*. This study was in agreement with previous findings where the key factors for water vole habitat selection is the availability of dense vegetation for food and cover (Lawton & Woodroffe, 1991; Aars *et al*. 2001; Telfer *et al*. 2001; Lambin *et al*. 2004; Fischer *et al*. 2009). The high percentage of fossorial breeding colonies and high water vole densities estimated from CMR indicate that grasslands are good quality habitat and an important habitat type for water vole populations in the area.

However, the findings of this study call into question the idea of water voles in the UK being riparian "specialists". Barreto *et al.* (1998a) proposed the Tightrope Hypothesis as an explanation for the decline of water voles in the UK: water voles continually walk a tightrope between survival and extinction because habitat loss and fragmentation has restricted their distribution to a narrow ribbon of riparian habitat. The additive effect of any other stochastic event, such as mink predation, is enough to make them "fall" off the tightrope (MacDonald & Harrington, 2010). The grasslands of the East End of Glasgow are a far cry from the pristine image of wetland habitat we associate with water voles; yet populations here appear to occur at some of the highest density ever recorded in the UK to the best of our knowledge. Even with the slowing of the decline in Britain reported by Strachan (2004), the East End populations represent a glimmer of hope for water voles: not only are these populations immune to mink predation given their distance from water courses, grassland habitat is commonplace, easier and cheaper to create and capable of supporting much higher numbers compared to riparian habitat. The city of Glasgow alone hosts over 90 parks and green spaces (Glasgow City Council, 2015). Whether the occurrence of fossorial behaviour is unique to Glasgow populations of water vole is a question still to be addressed yet grassland populations here appear to be bucking the trend of decline. It is possible that the limitations of standard surveying techniques has resulted in grassland populations being missed throughout the UK.

Grasslands should be considered a refuge for water voles, similar to reed beds but with the huge benefit of being a product of urbanisation rather than one under threat from it. While a longer-term study is required to further understand metapopulation dynamics and life history of the East End water voles, the occurrence of these populations does highlight the need for a re-think on what is suitable habitat for water voles. Grasslands could be a critical management tool for the conservation of this protected species.

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APPENDIX 1: Water Vole Survey Form Example (page 1)

WATER VOLE SURVEY FORM

SITE NAME Cranhill. Park
SITE NUMBER 31.1
GRID REF 644655
RECORDER RS.
DATE 17.9.14
SURVEY DISTANCE P/A. 100m

HABITAT INFORMATION

HABITAT:
- PARK/GARDEN ✓
- ROAD VERGE
- RANK GRASSLAND
- DERELICT
- DITCH
- OTHER

ASPECT: 229°

BANK:
- Boulders
- Stones
- Gravel
- Sand
- Earth ✓
- Reinforced

BANK PROFILE 5°

BORDERING LAND USE:
- Housing ✓
- Industrial
- Park/garden ✓
- Road verge
- Broadleaf woodland
- Conifer wood
- Permanent/temporary grassland
- Fencing
- Marsh
- Other

VEGETATION:
- Trees
- Bushes
- Herbaceous
- Reeds/sedges
- Weeds
- Tall grass ✓
- Short grass ✓

WILDLIFE INFORMATION

WATER VOLE FIELD SIGNS:
- SIGHTINGS
- TUMULI ✓
- LATRINES ✓
- BURROWS ✓
- RUNS ✓
- FEEDING REMAINS ✓
- LAWN ✓
- FOOTPRINTS

RAT
~~Sightings~~
~~Droppings~~
~~Footprint/run~~

FOX
~~Sightings~~
~~Droppings~~
~~Footprints~~

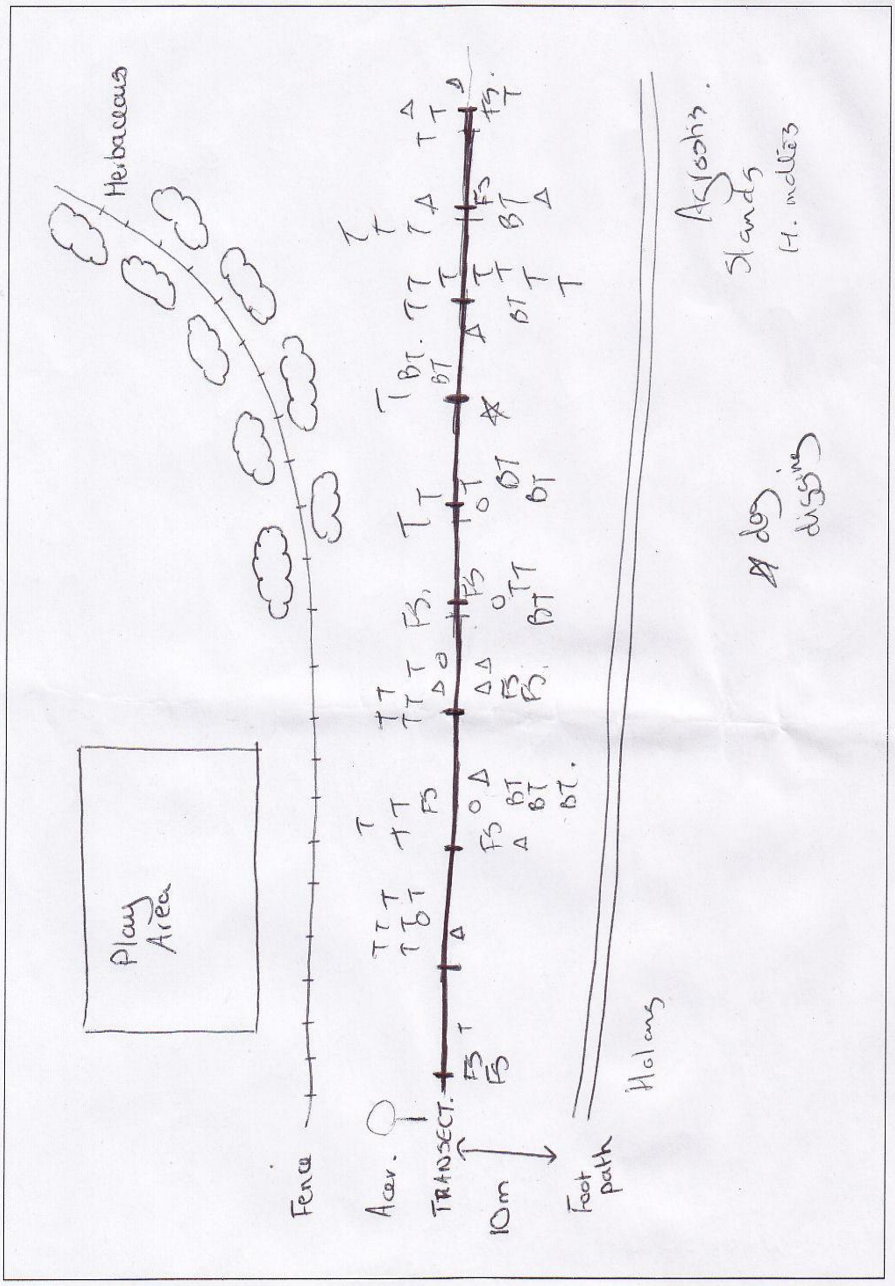
CAT
~~Sightings~~
~~Droppings~~
~~Footprints~~

OTHER WILDLIFE: High Disturbance
Dog.

PLANTS IDENTIFIED FROM FEEDING REMAINS:
H. lanatus
Agrostis

APPENDIX 1: Water Vole Survey Form (page 2)

SKETCH OF SITE



ADDITIONAL COMMENTS:

H. lanatus
Agrostis
H. mollis

Rumex.
Perennial rye grass.
Dandelion.
C. buttercup.

M. vetch

FS - Feeding station
T - Tumulus
O - Burrow
BT - Blocked tunnel
Δ - Nest material

Site Number	Date	Site Name	Grid OS
1.1	21.03.14	Royston road verge	632674
1.2	21.03.14	Robroyston Railway Bridge	633677
3.1	09.04.14	Millerston - Bogside road	642681
3.2	09.04.14	Millerston - field	645684
4.1	10.03.14	Hogganfield grassy hill	646675
4.2	10.03.14	Hogganfield reedbed	647673
5.1	10.03.14	Lethamhill Golf Course	645665
5.2	14.03.13	Hogganfield hill	644668
6.1	27.03.14	Frankfield housing estate	657681
6.2	27.03.14	Stepps - Whitehill farm	653689
7.1	10.03.14	Cardowan open field	653671
7.2	10.03.14	Strathclyde playing fields	654676
8.1	28.02.14	M8 embank Garthamlock	657663
8.2	06.03.14	Queenslie M8 embank	657661
9.1	24.03.14	Stepps road verge	663691
9.2	09.04.14	Stepps - Glen Plantation	667694
10.1	19.03.14	Buchanan Business Park	669688
10.2	27.03.14	Buchanan Business Park - railway	667685
11.1	24.03.14	Cardowan footpath	669679
11.2	24.03.14	Cardowan field verge	666679
12.1	06.03.14	Fort green wall	663662
12.2	06.03.14	Auchinlea park	666664
13.1	19.03.14	Crow wood	671692
13.2	19.03.14	Crow wood open field	673696
14.1	27.03.14	Garnkirk - forest footpath	673684
14.2	27.03.14	Garnkirk - railway	679684
15.1	24.03.14	Gartloch open field	672675
15.2	24.03.14	Gartloch field verge	675677
16.1	12.03.14	Craigend fields	673665
16.2	12.03.14	Craigend fields	678668
17.1	11.03.14	Easterhouse Kildermorie road	670657
17.2	11.03.14	Easterhouse park	678653
18.1	19.03.14	Cryston open field	688703
18.2	02.04.14	Bridgend footpath	689708
19.1	19.03.14	Cryston Sport Centre	689698
19.2	11.04.14	Muirhead - Electricity pylon	688694
20.1	26.03.14	Heathfield Moss	684689
21.1	26.03.14	Gartloch village	687673
21.2	26.03.14	Gartloch cottages - Bishop Loch	689671
22.1	11.03.14	Lochwood - Lochdochart Road	688663
22.2	11.03.14	Bishop Loch	688666
23.1	11.03.14	West Maryston FC woodland	685652
23.2	11.03.14	West Maryston rank grassland	689654
24.1	02.04.14	Moodiesburn - Gartferry road	692712
24.2	02.04.14	Moodiesburn - bowling	693716
25.1	02.04.14	Moodiesburn - Bedlay castle	694701
25.2	02.04.14	Moodiesburn - gas pipe	691705
26.1	26.03.14	Glaudhall Farm	693692
26.2	02.04.14	Muirhead A80	691698

27.1	26.03.14	Johnston Loch	698689
27.2	11.04.14	Muirhead - Mount Ellen	694690
28.1	26.03.14	Gartcosh M73	698676
28.2	26.03.14	Gartloch - Lochview cottages	692676
29.1	11.03.14	Lochend wood FC	693662
29.2	11.03.14	Lochwood fields	692664
30.1	08.04.14	M8 - Heatheryknowe	698654
30.2	08.02.14	Commonhead road	692658
31.1	28.02.14	Cranhill park	644657
31.2	08.04.14	Strone Road	649655
32.1	08.04.14	Toward Road	651655
32.2	08.04.14	Queenslie car park (no access)	658659
33.1	06.03.14	Queenslie M8 embank	664659
33.2	11.04.14	Wellhouse Cresent - play area	666654
34.1	12.03.14	Robroyston Park	628681
34.2	12.03.14	Robroyston Park	628684

Habitat type	Habitat category	Overall P/A Score	Distance from water/m
Road verge	Road verge	1	52
Road verge	Road verge	0	625
rank grassland	Rank grassland	0	526
rank grassland	Rank grassland	0	51
Park/garden	Park	1	162
Park/garden	Park	1	132
Park/garden	Park	0	185
Park/garden	Park	1	816
Road verge	Road verge	0	510
Rank grassland	Rank grassland	0	50
Park/garden	Park	1	161
Other	Park	0	250
Road verge	Road verge	0	1012
Road verge	Road verge	1	1148
Field verge	Rank grassland	0	148
Road verge	Road verge	0	87
Rank grassland	Rank grassland	0	50
Rank grassland	Rank grassland	0	470
Field verge	Rank grassland	0	284
Field verge	Rank grassland	0	189
Road verge	Road verge	1	50
Park/garden	Park	0	336
Field verge/grass strip	Agricultural	0	23
Open field/permament gra	Agricultural	0	222
Open field/rank grassland	Rank grassland	0	96
Open field/rank grassland	Rank grassland	0	345
Rank grassland	Rank grassland	1	239
Rank grassland	Rank grassland	1	50
Permanent grassland	Rank grassland	0	45
Permanent grassland	Rank grassland	0	383
Road verge	Road verge	1	70
Park/garden	Park	0	643
Rank grassland	Agricultural	0	810
Road verge	Road verge	0	298
road verge	Road verge	0	50
Field verge	Agricultural	0	50
Heath/permament grasslan	Rank grassland	0	357
Field verge	Agricultural	0	105
Rank grassland	Agricultural	1	66
Rank grassland/derelict	Vacant land	0	332
Grassland/marsh	Park	1	54
Road verge	Road verge	0	70
Rank grassland/derelict	Rank grassland	0	283
Openfield/rank grassland	Rank grassland	0	179
Open field/rank grassland	Rank grassland	0	147
Open field/rank grassland	Rank grassland	0	50
Open field/rank grassland	Rank grassland	0	50
Field verge	Agricultural	0	173
Field verge	Agricultural	0	50

Rankgrassland	Rank grassland	0	377
Field verge	Rank grassland	0	78
Rank grassland	Rank grassland	0	75
Rank grassland	Rank grassland	0	154
Permament grassland	Park	0	300
Agricultural/grassland	Agricultural	0	50
Road verge	Road verge	0	50
Derelict	Vacant land	0	50
Park/garden	Park	1	567
Derelict (playing field)	Vacant land	1	219
Derelict	Vacant land	1	754
Industrial	Road verge	1	518
Road verge	Road verge	1	585
Park/garden	Park	1	1126
Park/garden	Park	1	45
Park/garden	Park	1	70

Altitude/m	Profile	Aspect	Burrows/runs	Tumuli	Blocked tunnels	Lawn/feeding signs
88	10	248	3	0	0	1
89	20	338	0	0	0	0
90	flat	0	0	0	0	0
63	<3	198	0	0	0	0
97	15	269	0	0	0	1
82	<5		3	0	2	0
87	<3	220	0	0	0	0
95	<3	330	1	0	0	0
88	25	71	0	0	0	0
90	<3	172	0	0	0	0
88	10	358	1	0	2	0
95	<25	348	0	0	0	0
95	<3	187	6	2	0	0
75	<3	356	4	1	0	0
93	3	356	0	0	0	0
85	5	256	0	0	0	0
89	5	185	0	0	0	0
90	<3	347	0	0	0	0
82	3	242	0	0	0	0
95	<3	177	0	0	0	0
75	50	200	3		10	8
89	flat	N/A	0	0	0	0
93	flat	N/A	0	0	0	0
86	<3	12	0	0	0	0
93	10	352	0	0	0	0
97	flat	N/A	0	0	0	0
80	<3	208	0	0	0	1
80	flat	N/A	0	0	0	1
91	10	143	0	0	0	0
75	<3	40	0	0	0	0
75	10	228	2	2	3	1
75	<25	354	0	0	0	0
76	10	343	0	0	0	0
79	5	91	0	0	0	0
83	5	127	0	0	0	0
100	<3	348	0	0	0	0
90	flat	N/A	0	0	0	0
83	<3	5	0	0	0	0
80	10	156	5	2	0	3
92	10	36	0	0	0	0
84	3	253	0	0	1	2
84	5	210	0	0	0	0
81	0	N/A	0	0	0	0
66	5	332	0	0	0	0
69	3	208	0	0	0	0
83	flat	N/A	0	0	0	0
65	flat	N/A	0	0	0	0
85	5	169	0	0	0	0
74	5	333	0	0	0	0

87	5		138	0	0	0	0
90	flat	N/A		0	0	0	0
80	0	N/A		0	0	0	0
80	<10		185	0	0	0	0
87	0	N/A		1	0	0	0
84	<10		200	0	0	0	0
88	20		304	0	0	0	0
83	<5		298	0	0	0	0
	<5		229	4	2	2	0
84	flat	N/A		5	3	0	5
83	flat	n/A	Burrows/lawns/feeding remains - transect not done due to <100m				
76	25		354	5	4	0	5
90	<3		23	1	0	2	0
68	10		194	2	2	0	0
92	flat	N/A		2	2	2	0
80	5		350	6	0	4	6

Droppings	Latrines	Distance category	Overall P/A Score	Vegetation type
0	0	0-150m	1	Tall grass/weeds(bramble)
0	0	>550m	0	Tall grass/weeds
0	0	151-550m	0	tall grass/weeds
0	0	0-150m	0	tall grass/short grass
0	0	151-550m	1	tall grass/short grass/reed
0	0	0-150m	1	Reed/tall grass/trees
0	0	151-550m	0	Tall grass/reed/moss
0	0	>550m	1	Tall grass
0	0	151-550m	0	tall grass/short grass/juncus
0	0	0-150m	0	Juncus/short grass
0	0	151-550m	1	tall grass/reed
0	0	151-550m	0	tall grass
0	0	>550m	0	tall grass
0	0	>550m	1	tall grass/moss/reed
0	0	0-150m	0	juncus/tall grass
0	0	0-150m	0	short grass/clumps tall grass/weeds
0	0	0-150m	0	tall grass/short grass
0	0	151-550m	0	tall grass/short grass
0	0	151-550m	0	short grass/bushes
0	0	151-550m	0	Tall grass/Juncus/trees
2	0	0-150m	1	tall grass/short grass
0	0	151-550m	0	tall grass/short grass
0	0	0-150m	0	tall grass/juncus/trees
0	0	151-550m	0	tall grass/juncus
0	0	0-150m	0	short grass/Juncus
0	0	151-550m	0	tall grass/juncus
0	0	151-550m	1	Tall grass/juncus
0	0	0-150m	1	Tall grass/Juncus
0	0	0-150m	0	tall grass/thistles
0	0	151-550m	0	tall grass
1	0	0-150m	1	tall grass/short grass
0	0	>550m	0	tall grass/daffodils
0	0	>550m	0	tall grass/hawthorn trees
0	0	151-550m	0	tall grass/weeds/trees
0	0	0-150m	0	tall grass/juncus/trees
0	0	0-150m	0	juncus/tall grass/short grass
0	0	151-550m	0	short grass/club moss/heather
0	0	0-150m	0	reeds/short grass
0	0	0-150m	1	Juncus/short grass
0	0	151-550m	0	tall grass/gorse
0	0	0-150m	1	tall grass/Juncus
0	0	0-150m	0	tall grass/bushes/bramble
0	0	151-550m	0	tall grass/reeds/trees
0	0	151-550m	0	tall grass/juncus/weeds
0	0	0-150m	0	tall grass/short grass/weeds
0	0	0-150m	0	short grass/tall grass/moss
0	0	0-150m	0	tall grass/short grass/juncus
0	0	151-550m	0	tall/short grass/weeds/reeds/trees
0	0	0-150m	0	short grass/juncus

0	0	151-550m	0	tall grass/Juncus
0	0	0-150m	0	Juncus/short grass/tall grass/weeds
0	0	0-150m	0	tall grass/Juncus
0	0	151-550m	0	tall grass/short grass/weeds/trees
0	0	151-550m	0	tall grass/reeds
0	0	0-150m	0	tll grass/short grass
0	0	0-150m	0	tall grass/short grass/trees
0	0	0-150m	0	short grass/tall grass
0	0	>550m	1	tall grass
0	0	151-550m	1	short grass/tallgrass
		>550m	1	
0	0	151-550m	1	short grass/tall grass/weeds
0	0	>550m	1	tall grass/herbaceous
0	0	>550m	1	short grass/grass just been cut
0	0	0-150m	1	tall grass
0	0	0-150m	1	Juncus/tall grass

Vegetation height	Predator signs	Disturbance
>1.5m	0	Moderate
50	Fox	Low
1.25m	Fox	Low
1.25;30	Fox	Low
30	0	Moderate
30	Fox	Moderate
30	Fox	Moderate
30	0	Moderate
30;grass clumps 1m	Fox	High
80	0	Low
50	Fox/Cat	High
25	Fox/Cat	Moderate
30	Fox	Moderate
30	0	Moderate
80	0	Low
mix heights	0	Moderate
30	0	Low
30	Fox	Low
30	Fox	Low
1.5	0	Low
30	Fox/cat	High
30	0	Moderate
80	Fox	Low
1.25	Fox	Low
30	Cat	Low
1m	Rat	Low
80	Fox	Low
50	0	Low
50	Fox	Low
>1m	Fox	Low
80 between 1-6;20 after	0	Moderate
50	fox	Moderate
80	cat/fox	Low
80	0	Low
80	0	Low
50 Juncus;mix heights	0	Low
30	0	Low
50	0	Low
50 for 5 (Juncus)/80 tall for 5	0	Low
50	0	Moderate
30	0	Low
1m bramble ticket 20m length	0	Low
80	Fox/cat	Low
1.25	0	Low
50	0	Low
30	Fox	Low
50	Cat	Moderate
mix heights:	Cat	Low
grass 30;clumps juncus 50	0	Low

tall grass 1.5/juncus 50cm			0	Low
mix heights		Fox/cat		Low
	50		0	Low
mix heights: 30/1.25m grass/bramble thicket 50cm		fox/Cat		Low
Juncus closest to river;tall grass			0	Low
	50		0	Low
grass 30;clumps tall grass 80		Fox		Low
grass 30;clumps tall grass 80			0	Moderate
	30		0	Moderate
grass 30;clumps tall grass 80			0	Moderate
				Moderate
grass 30; clumps 80			0	Low
	30		0	Moderate
	5	Fox		Low
	50	Fox		Moderate
	50		0	Low

Land use	Neighbouring land categories	Additional
Housing	Park	Roe/boggy ground at bottom of th
Housing/Industrial	Park	Roe/Field vole/earth compacted w
Industrial/broadleaf woodland	Road	Digging at 2 points by fox/dog; ext
Marsh/agricultural	Natural	Fox den nearby; WV signs at locha
Park	Park	
Park/reedbed	Park	
Golf course/road verge	Park	Ground boggy with numerous tall
park	Park	Beside hawthorn hedge
Housing/industrial/marsh	Park	Ground compacted; Herpetosure f
Agricultural/road verge	Natural	Field vole signs
Housing/LNR	Park	
Other/LNR	Natural	
road verge/housing	Road	High occurrence of dog faeces
Road verge/industrial	Road	Extenisve burrows in grassland bet
Road verge/agricultural	Road	Ground boggy/feeding signs of FV
Road verge/agricultural	Road	Ground impacted/hard; WV signs a
Industrial/marsh	Natural	No signs in grassland but feeding s
Railway/industrial	Road	Field vole signs;ground compacted
Broadleaf woodland/housing	Natural	Ground felt compacted/over-shad
Marsh/agricultural	Natural	Mole/Field vole signs
road verge/industrial	Road	burrows spread out to flat area be
park	Park	Field vole signs
Agricultural	Natural	Field vole/2 large Ash trees/fox ru
Agricultural	Natural	Field vole/badger
Railway verge/broadleaf	Natural	Rabbit burrows; ground compacte
Railway verge/rank grassland	Natural	Rat feeding station (grasses, snail s
Reedbed/broadleaf wood	Natural	Feeding signs found in Juncus stan
Tall grass/marsh	Natural	Feeding signs closest to river/Mole
Broadleaf woodland/housing	Natural	Digging at 3 points
Broadleaf woodland/housing	Natural	Field vole signs
road verge/housing	Road	Dogs/litter
road verge/park	Road	Field vole/rabbit signs
Housing/agricultural	Road	Field vole/small mammal
Housing/agricultural	Park	Field vole; ground compacted with
Industrial (sport centre)	Road	Dogs/Field vole signs/molehills
Agricultural	Natural	Extensive Field vole signs
Broadleaf woodland	Natural	Roe
Road verge/agricultural	Natural	Roe/land churned up by livestock
Reedbed/agricultural	Natural	Field volesigns in long grass sectio
housing	Park	Dogs/Field vole
Reedbed/agricultural	Natural	15m distance from reedbed/roe/g
broadleaf woodland	Natural	Field vole/roe deer/ Small torto
Agricultural/housing	Road	Field vole/mice signs prevalent
Industrial/reedbed	Natural	Mole; Field vole; evidence of diggi
Marsh	Natural	205
Road verge/broadleaf	Road	Digging by fox/Field vole signs
Housing/park	Park	Field vole signs; floodplain of river
Agricultural/road verge	Natural	Boggy ground/overshading by tree
Agricultural/road verge	Road	Transect started from Ash tree

Housing/road verge	Road	
Housing/agricultural	Road	Fox remains found; Field vole signs
Road verge/reeds	Natural	Bothlin burn 20m distance
Housing/agricultural	Natural	Roe/ground earth with rubble
broadleaf woodland	Natural	Molehills over 4 points; river 40m
Agricultural	Natural	Molehills over 6 points;rabbit burr
Agricultural	Natural	Ground felt compacted
housing/broadleaf wood	Park	ground felt compacted; developm
park/housing	Park	Dog walkers
road verge/housing	Road	Next to St Maria Gorettit PS; burro
Housing	Road	
Industrial	Road	No access but signs clearly visible t
road verge/industrial	Road	
Housing/derelict	Road	Grass recently cut - damage to bur
park/reedbed	Park	Dog walkers; 25m from pond
reedbed/housing	Park	Dogs;distance from reedebed 25m

ıe road verge at foot of houses
/ith some rubble
:ensive Field vole signs; looks favourable for WV
n 25m away

trees (willow)

-ence running parallel to road

tween road and fence
in Juncus strip(3m)/Roe/horses
along riverbank of Garnkirk burn
igns in the marshy area to west of transect
l with some rubble;fox den 10m from transect
ing by trees

yond fence onto motorway verge

n through strip

d
shells)
d which follows a ditch from Gartloch pond/Roe
ehills across 3 points/Field vole signs towards drier section

ı some rubble

n

round churned up by cattle
iseshell

ng by predator

- 26m distance
es

s

distance from transect
rows;Field vole signs

ent going on nearby

ows evident under concrete block at point 7; more burrows towards Skerryvore road

through fence (distance <5m)

rrow entrances; WV in slope behind shops too

n

Cranhill Park Trapping 11-15th August 2014

Site	Cranhill			
Date		12.08.14	12.08.14	12.08.14
Trap number		5(T2)	9(T3)	12(T4)
Trap type		S	S	S
Time of capture		2.00pm	2.00pm	2.00pm
AVID		030 874 835	030 845 576	031 031 000
Mass (g)		108.3	97.4	118.5
Body length (mm)		135	149	151
Tail length (mm)		90	85	100
Hind foot length (mm)		28	29	30
Anal-genital gap (mm)		8	8	9
Colour		Black	Black	Black
Sex		Female	Female	Female
Ectoparasites		No	Fleas	Fleas/mites
Additional notes			Chip reader failed because of rain. Hair clip left tail base as precaution	
		Rain	Rain	Rain
Date of recapture				
Time				
Trap number				
Trap type				
Mass (g)				

12.08.14	12.08.14	12.08.14		14.08.14
13(T5)	14(T6)	16(T8)		14 (T6)
T	S	T		S
2.00pm	2.00pm	2.00pm		5.30am
031 021 079	031 041 592	Not chipped	No check on 13th Robyn ill. Traps set at 8pm.	031 091 260
111.3	105.8	122.1		106.7
156	151	152		144
100	84	106		95
31	30	32		32
13	10	16		10
Black	Black	Black		Black
Male	Female	Male		Female
	Fleas, dander	Fleas		
Rain	Rain	Went into shock on handling. Hair clip 2 patches - LH and RH tail base. Fast release.		
	15.08.14			14.08.14
	6.30am			1.00pm
	14 (T6)			14(T6)
	104			S
				107.1

14.08.14	14.08.14	14.08.14	15.08.14	15.08.14
20(T9)	2 (T1)	13(T5)	5(T2)	12(T4)
S	S	S	T	S
5.30am	1.00pm	1.00pm	1.00pm	1.00pm
031 015 806	030 887 042	030 876 609	Not chipped - stress	Not chipped - stress
86.2	104.8	110.4	91.6	108.8
150	152	142	144	173
85	93	91	80	109
30	30	29	29	33
9	9	8	9	8
Black	Brown	Black	Black	Brindle
Female	Female	Female	Female	Female
Lots fleas. Scruffy coat, ok body condition		Fleas	Fleas	Good body condition

15.08.14
16(T8)
S
1.00pm
031 008 871
98.2
146
95
31
8
Black
Female
Dander

Cranhill (2) Fieldsign Transect

Date:	20.08.14			
Transect number		1	2	3
Trap number		2	5	9
Fieldsigns				
Burrow		3	0	2
Blocked tunnel		5	4	4
Tumulus		3	3	3
Feeding station		2	5	1
Droppings		0	2	0
Latrine	1(old)		0	1
Expelled nest material		2	0	2
Other			Wasp nest in burrow	
Disturbance				
Human				
Predator				
Vegetation				
Sward height (average)	20(no tussocks)	25(no tussocks)	30(no tussock)	
Species diversity		3	3	3
Grass species	Holcus lanatus (90%) Holcus mollis (5%)	Holcus lanatus (80%) Holcus mollis (5%)	Agrostis sp. (75%) Holcus lanatus (15%)	
Feeding station spp.	Epilobium montanum (1)	Ranunculus repens (1) Hay from traps (Meadow foxtail)	Plantago lanceolata (5)	

4	5	6	7
12	13	14	15
1	2	6	2
2	8	4	4
2	0	5 (fresh)	1
2 fresh;2 old	1 old	3	3
0	0	0	0
1	1	0	1(old)
1	2	0	2
35 (dense,tussock)	35 (dense;tussock)	35(dense;tussock)	35 (dense;tussock)
3	2	2	2
Holcus lanatus (90%)	Holcus lanatus (85%)	Holcus lanatus (60%)	Holcus lanatus (80%)
Agrostis sp (10%)	Agrostis sp. (15%)	Agrostis sp (40%)	Agrostis (20%)
Plantago lanceolata (1)			

	8	9	10
	16	20	25
	1	4	2
	4	1	1
	2	1	2
	1	2	1 old
	0	0	0
	0	0	0
	1	0	2
35(dense;tussock)	30 (no tussocks)	20 (no tussock)	
	2	2	2
Holcus lanatus (90%)	Holcus lanatus (60%)	Agrostis (70%)	
Agrostis (10%)	Agrostis (40%)	Holcus lanatus (30%)	

Cranhill NVC Quadrats

Date	20.08.14				
GPS					
OS Grid		644655			
Altitude					
Slope aspect	182°				
Slope angle	5°				
Quadrat		1			2
Transect point		1			6
Area	2m x 2m			2m x 2m	
Species diversity		4	Domin scale		2
Species	Holcus lanatus		9 (76-90%)	Holcus lanatus	
	Holcus mollis		4 (4-10%)	Agrostis	
	Epilobium montanum		1 (<4% few)		
	Ranunculus repens		1 (<4% few)		

Soil analysis

pH		6.5		6.5
Nitrate	0mg/l		0mg/l	
Phosphorus	<5mg/l		<5mg/l	

		3	
		10	
	2m x 2m		
Domin scale		2	Domin scale
8 (51-75%)	Agrostis		8 (51-75%)
7 (34-50%)	Holcus lanatus		6 (26-33%)
		6.5	
	0mg/l		
	<5mg/l		

APPENDIX IV Summary Table of Environmental Variables: Trapping Sites

Trapping Site	**Alti-tude (m)**	**Aspect (º)**	**Slope profile (º)**	**Habitat type**	**Neighbour-ing land use**	**Soil Type**	**Soil pH**	**Disturb-ance level**	**Predator signs**	**Dominant grass species**	**Water vole population estimates from CMR**
Avenue End Primary School	85	263 (west)	10	Park/ garden	Road verge	Brown earth/ clay	Sample 1: 7.0 Sample 2: 6.5 Sample 3: 6.5	Low	Cat Fox	*Festuca rubra*	None trapped
Garthamlock Water Towers	101	77 (east)	20	Park/ garden	Road verge	Brown earth/ clay	Sample 1: 6.0 Sample 2: 6.5 Sample 3: 7.0	Low	Fox	*Holcus* species	None trapped
Cranhill Park	60	182 (south)	3	Park/ garden	Housing	Brown earth	Sample 1: 6.5 Sample 2: 6.5 Sample 3: 6.5	High	None	*Holcus* species	Trapping 1: 78 Trapping 2: 48
Tillycairn Drive	99	342 (north)	25	Park/ garden	Housing	Brown earth	Sample 1: 7.0 Sample 2: 6.0 Sample 3: 5.5	Mod	Fox Cat	*D. glomerata/ H. lanatus*	10