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Infilling wetlands with Construction and Demolition (C&D) waste: Influence on land use, plant / dipteran communities and metal contamination

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Abstract

Construction and demolition (C&D) waste, comprised of sand, stone, concrete, bitumen and other wastes from building sites, is one of the largest fractions of waste produced internationally. It frequently ends up being used for land reclamation (infilling) under specially granted permits in wetlands. This happens despite the instrumental value of wetlands to society, both environmentally and financially, in that they provide a plethora of vital ecosystem services including habitat provision, water regulation and filtration.

This study which assessed the distribution of C&D waste infill sites at a local scale, found that sites are primarily wetlands located adjacent to urban areas and major road networks. Wetlands in these areas are, therefore, likely to be at a higher risk of loss. Infill sites were also found to be concentrated around designated conservation sites (SACs), a point of concern as these habitats may be sensitive to any contamination and hydrological changes caused by the waste. The study also found that non-compliance with permit conditions is common, with many sites having excessive or contaminated waste; poor or absent perimeter fencing; infilling activities taking place prior to the granting of permits or after permits have expired. Undocumented infilling which was also found to be a major problem had a similar distribution pattern as legal infilling sites. Resources available to local authorities should be increased to allow better policing of proposed and current sites.

The ecological impacts of infilling wetlands with C&D waste were also assessed. It was found that plant species composition was different on the waste compared to the wetland, with an increase in common ruderal species and fewer wetland specialist species on the infill. This is most likely as a result of changed soil parameters where the pH increased and both soil moisture and organic content decreased. Dipteran communities were also found to differ, with a decrease in wetland specialist, gall-forming, parasitic and haematophagous groups. Both the abundance and species-richness of Marsh Flies (Sciomyzidae) were lower on the C&D waste infill than the adjacent wetland. In addition, slugs (*Deroceras reticulatum*) collected on C&D waste had significantly higher concentrations of priority pollutants Antimony, Arsenic, Barium, Cadmium, Cobalt, Selenium and

Thalium than those from control sites. This suggests that the metal in the C&D waste is in a bioavailable form, increasing the potential risk of such infill sites to adjacent wetland habitats, including those with European designations. Challenges faced through the study are discussed, and recommendations are made both for future research and policy makers.

Summary for Local Authorities and Policy Makers

Construction and demolition (C&D) waste, comprised of sand, stone, concrete, bitumen and other wastes from building sites, is one of the largest fractions of waste produced internationally. It frequently ends up being used for land reclamation (infilling) under specially granted permits in wetlands. This happens despite the instrumental value of wetlands to society, both environmentally and financially, in that they provide a plethora of vital ecosystem services including habitat provision, water regulation and filtration.

This study which assessed the distribution of C&D waste infill sites at a local scale, found that sites are primarily wetlands located adjacent to urban areas and major road networks. Wetlands in these areas are, therefore, likely to be at a higher risk of loss. Infill sites were also found to be concentrated around designated conservation sites (SACs), a point of concern as these habitats may be sensitive to any contamination and hydrological changes caused by the waste. The study also found that non-compliance with permit conditions is common, with many sites having excessive or contaminated waste; poor or absent perimeter fencing; infilling activities taking place prior to the granting of permits or after permits have expired. Undocumented infilling which was also found to be a major problem had a similar distribution pattern as legal infilling sites.

The ecological impacts of infilling wetlands with C&D waste were also assessed. It was found that plant species composition was different on the waste compared to the wetland, most likely as a result of changed soil parameters. Dipteran communities were also found to differ. In addition, slugs (*Deroceras reticulatum*) collected on C&D waste had significantly higher concentrations of priority pollutants Antimony, Arsenic, Barium, Cadmium, Cobalt, Selenium and Thalium than those from control sites. This suggests that the metal in the C&D waste is in a bioavailable form, increasing the potential risk of such infill sites to adjacent wetland habitats, including those with European designations (i.e. SACs).

Local authorities should be given the resources to implement an education programme for all permit applicants, to ensure they are cognisant of the environmental and legal consequences to breaking the conditions of their permit. The general public should also be made aware of the environmental consequences to infilling wetlands with C&D waste.

Local authorities also need sufficient resources to effectively monitor and police infill sites to ensure permit terms are adhered to. All applications made for infilling permits should require detailed ecological surveys (including botanical, invertebrates, vertebrates and hydrological) regardless of the site size or location and, where the site is within 15km of an SAC, a full AA should be undertaken. This will ensure that valuable wetland habitats are not lost. Permit terms should also include a stipulation that allows future environmental studies to be carried out on the sites, and if required (for assessing hydrological parameters), the installation of hydrological piezometers before completion of the infilling process. This initial survey (and AA) should also serve as a baseline for such studies, to effectively assess any impact that the C&D waste has had on the wetland. Lining C&D waste infill sites and treating the collected leachate would be the most effective method for minimising the contamination risk of the waste to groundwater.

Mitigation strategies should be built into each permit, ensuring that the most ecologically valuable and hydrologically sensitive areas of intact wetland, which may be locally important, remain as they are. This will ensure that their associated biodiversity and ecosystem services are not completely lost. For these sensitive areas, there is no 'safe' amount of C&D waste that can be used, as the ecological communities will likely change everywhere that is infilled.

In addition to EU and national waste policies, Local Authorities should have a clear strategy for C&D waste disposal. This should aim to reduce production and increase recycling rates through: educating the construction sector workforce; introducing financial incentives for low waste building methods such as prefabricated housing; taxation of raw materials and subsidies for recycled materials. In addition it should also ensure that the most environmentally sensitive areas in that local authorities' area are protected from infilling activities.

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A special thanks to my parents, Martin and Olive, who have given me unending encouragement and support through the project, as did my sisters Audrey and Emma and brother Martin, despite not seeing me very often! My sincere thanks to all my friends for the fun, adventure and support through this project.

Lastly, but certainly not least, I wish to thank my girlfriend, Elaine Williams, for her incredible support, love and patience through the project.

Dedication

I wish to dedicate this thesis to the memory of my sister, Carmel Staunton. Carmel was a fun, caring and talented young woman, who was tragically killed in a road accident on October 24th, 2003 at the age of 22. She had a passion for psychology, and completed her BSc in Psychology through Science in NUI Maynooth, obtaining 1st class honours. She had planned to return to University to pursue a PhD after taking a year out to focus on her other life passion, music. Although she never got to see them published, her final year undergraduate thesis resulted in her co-authoring two international peer-reviewed scientific papers, and being lead author on an Irish peer-reviewed scientific paper. Carmel, your ever-optimistic attitude and dedication to pursuing your passions and living life to its full was truly an inspiration.

Chapter 1:

General Introduction

1. General Introduction

1.1. Background

The ‘building boom’ associated with the period of economic growth during the mid 2000s (ESRI, 2014) resulted in increased volume of construction and demolition waste being produced in Ireland (Chapter 2, Fig 1) and Europe (Schrör, 2011). The waste can be quite heterogeneous, and varying regionally or depending on the activity being carried out (construction or demolition) and the building style (concrete, timber, etc.) (Fischer and Werge, 2009). Historically, the waste was either disposed of in municipal landfills or, due to its perceived inertness, was used as unregulated fill material locally (Symonds Group Ltd., 1999). Currently, its disposal is generally permitted in unlined landfills (often on wetlands to help in land reclamation) throughout Ireland (Duran *et al*., 2005) and the world (Poon *et al*., 2004; Symonds Group Ltd. *et al*., 1999; USEPA, 2009).

1.2. Construction and demolition waste

Construction and demolition (C&D) waste is waste that is “generated when new structures are built and when existing structures are renovated or demolished (including deconstruction activities)” (USEPA, 2009). The actual composition of the waste varies from one country to the next depending on building practises (Franklin Associates, 1999; Hyder Consulting *et al*., 2011; Poon *et al*., 2001; Symonds Group Ltd. *et al*., 1999) and economic activity (Fischer and Werge, 2009), but it generally includes concrete, brick, asphalt/bitumous mixtures, timber, gypsum, metals, plastics, ceramics, soil and stones (Symonds Group Ltd. *et al*., 1999). The European Union (EU) has classified 28 sub-categories of C&D waste, of which 16 are known to be hazardous (such as C&D wastes containing polychlorinated biphenyls, coal tar and tarred products) (EPA, 2002). The inert fractions of the waste (i.e. soil, stones, concrete, brick, etc.) are often disposed of in unlined landfills or used for land reclamation following separation from any hazardous material (i.e. metals, gypsum, paints, etc.) (Symonds Group Ltd. *et al*., 1999; Schrör, 2011; Fischer and Werge, 2009). Separation should occur using crushers (such as jaw or impact crushers) followed by removal of contaminants (metals, plastics, gypsum, etc.) with electromagnets and manual separation (Symonds Group Ltd. *et al*., 1999). However, this separation is often not fully

effective, so some hazardous waste inevitably remains with the inert fraction that goes to be used as infill material (Roussat *et al*., 2008).

Large amounts of C&D waste are produced globally each year, and it was the largest (32.9%) waste fraction in the EU in 2008 (latest data), when 859 million tonnes were produced (Schrör, 2011). Available data from Ireland can be seen in Chapter 2 (Fig. 1). However, there appears to be confusion regarding the constituents of C&D waste internationally. The latest data show the US have excluded waste from natural disasters and roads (2003 data which showed production of 170 million tonnes; USEPA, 2009) while Australia excluded excavated material, soil and stones (data showed 19 million tonnes produced between 2008-2009; Hyder Consulting *et al*., 2011). Even within Europe this confusion occurs, with countries having different (or an absence of) information on the constituent of C&D waste for their respective reports, resulting in poor quality data for any international comparisons (BIO Intelligence Service, 2011).

In addition to the variation in constituents of C&D waste, there are many variations and uncertainties in the estimations of production and disposal/recycling rates (BIO Intelligence Service, 2011; USEPA, 2009). This is mainly due to a lack of reliable data (many C&D waste infill sites are self regulated so may be open to bias) and the nature by which the data are collected from a mixture of actual data, questionnaires and estimations (BIO Intelligence Service, 2011; Hadjieva-Zaharieva *et al*., 2003; Schrör, 2011; USEPA, 2009; Wu *et al*., 2014). To further add to the confusion for C&D waste recycling data, the term ‘recycling’ is often used in National datasets (for European countries) to describe both ‘waste recycling’ and ‘waste recovery’ (which includes infilling or land reclamation; Council Directive, 2008/98/EC) (Fischer and Werge, 2009; Tojo and Fischer, 2011).

The use of C&D waste for land reclamation or infilling is deemed by EU legislation (Council Directive, 2008/98/EC) to fall under the term recovery (without energy recovery), defined therein as a process which results in “waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfil a particular function”. While infilling activity would be unlikely to halt completely in the absence of C&D waste, it has been suggested it

would be unlikely to occur to the same extent (Symonds Group Ltd. *et al*., 1999). Recycling C&D waste, a more suitable alternative to recovery, would involve the reprocessing of the waste for use, according to the European Council Directive (2008/98/EC). There has been extensive research undertaken regarding the potential for and viability of recycling C&D waste (Bianchini *et al*., 2005; Coelho and de Brito, 2013; Duran *et al*., 2006; Hiete at al., 2011; Lawson and Douglas, 2001; Rao *et al*., 2007; Tomas *et al*., 1999; Weil *et al*., 2006; Yuan *et al*., 2011; Zhao *et al*., 2010), with many of these studies focusing on the specific method development for treating waste so it can be used in high value end products, such as concrete (Bianchini *et al*., 2005; Rao *et al*., 2007; Tomas *et al*., 1999; Weil *et al*., 2006). One of the major factors currently preventing effective C&D waste recycling is the cost associated with it (Duran *et al*., 2005; Yuan *et al*., 2010). However, it has been found that the implementation of financial incentives (taxation on the use of virgin material combined with subsidies on recycled aggregates) would best encourage the use of recycled C&D waste by generating a market for the material (Duran *et al*., 2005; Yuan *et al*., 2010).

The most efficient method of reducing the amount of C&D waste sent to infill sites is by minimisation at source, on construction sites (Poon *et al*., 2004; Tam and Tam, 2008; Teo and Loosemore, 2001). Although some C&D waste production on-site is inevitable (Poon *et al*., 2013), some avoidable issues such as poor worker skill (e.g. wrong measurements) and poor transportation (e.g. insufficient protection around new materials) are thought to be responsible for wasting significant amounts of good building material (Yuan *et al*., 2011). Effective reduction on-site is best brought about by financial incentives (Tam and Tam, 2008; Yuan *et al*., 2011), rather than simply implementing legislation to reduce its production, which was shown to have poor results (estimated only 5% reduction in waste production) in Hong Kong (Poon *et al*., 2013). Reducing the production of C&D waste and increasing the rate of recycling not only reduces the need for disposal thereby infilling fewer wetlands, but it also has the added benefit of decreasing the extraction of virgin material (Hadjieva-Zaharieva *et al*., 2003).

Traditionally, most C&D waste should have been disposed of in municipal waste landfills (Esin and Cosgun, 2007; Symonds Group Ltd. *et al*., 1999). However, over the past few decades C&D waste has been often used under special permits

(See Section 2.1 and Chapter 2, Fig 2) for infilling sites in unlined landfills. These C&D waste infill sites are usually directly regulated or overseen by local authorities and governmental organisations (Poon *et al*., 2004; Symonds Group Ltd. *et al*., 1999; USEPA, 2009), such as the EPA and Local Authorities in Ireland who grant the permits based on EU and national legislation (Council Directive 2006/12/EC; Council Directive 2008/98/EC; Council Directive 2008/105/EC; Statutory Instrument No. 10/1996; Statutory Instrument No. 165/1998; Statutory Instrument No. 821/2007). These infill sites are often located on wetlands with the aim of creating drier and more productive lands for agriculture both in Ireland (Duran *et al*., 2005) and internationally (Coelho and de Brito, 2013; Lawson and Douglas, 2001; Poon *et al*., 2013). Infilling has been described as "adding any material to raise the bottom level of a wetland or to replace the wetland with dry land" (Vottler and Muir, 1996).

1.3. Wetlands

Wetlands have been broadly defined by the Ramsar Convention (1971) as "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres". They have been estimated to cover from seven to over twelve million km^2 globally (Millennium Ecosystem Assessment, 2005; Mitsch and Gosselink, 2007; Fig 1.1). As this thesis will be dealing only with sites on inland/freshwater wetlands, a more suitable definition in this context may be ecosystems which "arise when inundation by water produces soils dominated by anaerobic processes which, in turn, forces the biota, particularly rooted plants, to adapt to flooding" (Keddy, 2010).

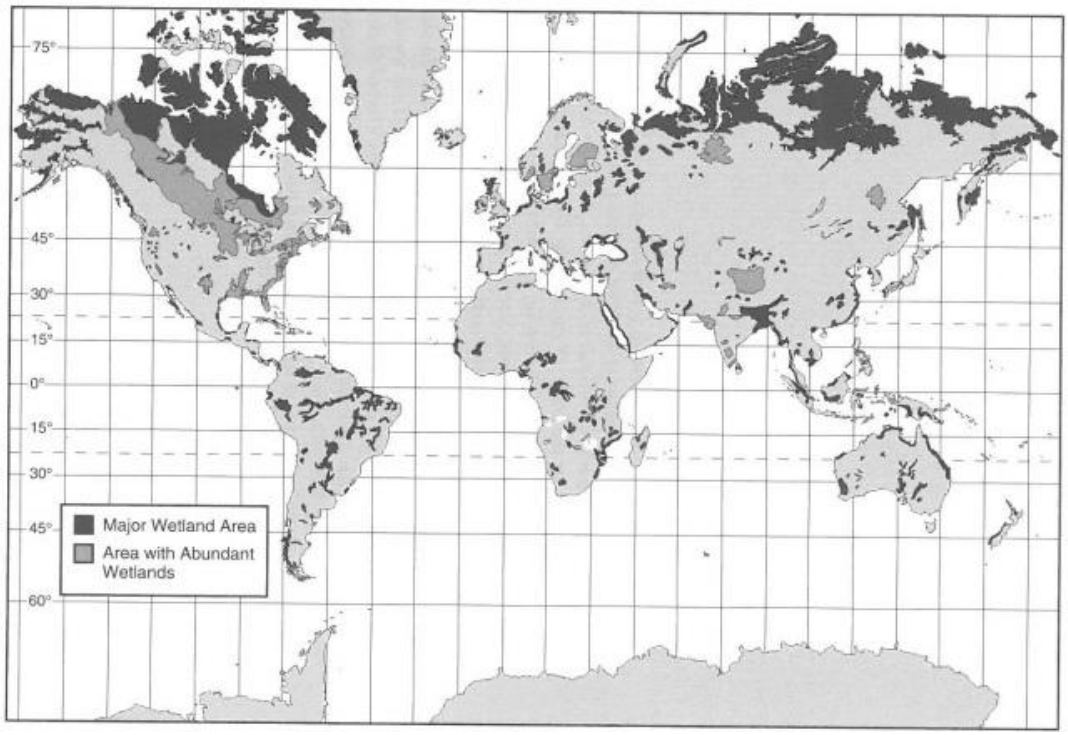

Fig 1.1. *Map showing the global distribution of major wetlands (Source: Mitch and Gosselink, 2007).*

Wetlands are important ecosystems, providing a plethora of ecosystem services that directly benefit humans, including (but not limited to) the cleansing of water, water supply stabilisation, flood protection, carbon sequestration, food and resource production, habitat provision and providing areas for recreational activities (Costanza *et al*., 1997; EC 1995; Keddy, 2010; Mitsch and Gosselink, 2007). A meta-analysis carried out by Brander *et al*. (2013) discussed the mean value of a variety of wetlands around the world relating to particular ecosystem services. Brander *et al*. found that, in general, wetlands were worth US$ 6,923 ± 3,186/ha/year for the their flood control services alone. Their contribution to increasing water quality, and providing a suitable water supply were estimated to be worth US$ 5,788 ± 3,131/ha/year and US$ 3,389 ± 2,015/ha/year respectively (Brander *et al*., 2013). The total monetary value, in socio-economic terms (incorporating all ecosystem services), specifically of inland wetlands has be estimated by de Groot *et al*. (2012) as being approximately US$ 25,682/ha/year. This estimate by de Groot *et al*. was further broken into provisioning (food, timber, water, etc.; US$ 1,659/ha/year), regulating (climate, water, and air quality regulation, nutrient cycling, etc.; US$ 17,364/ha/year), habitat (genetic diversity,

etc; US$ 2,455/ha/year) and cultural (recreation, etc.; US$ 4,203/ha/year) services. However, the total value of any individual wetland varies depending on factors including its size, the presence of other nearby wetlands, wetland type, condition, and nearby socio-economic activities (Brander *et al*., 2013). It has been shown that the monetary value (in socio-economic terms) of these ecosystem services associated with an intact wetland can far exceed the value of these improved lands (Balmford *et al*., 2002). The preservation of these valuable ecosystem services should therefore play an important role in the development of future environmental policy and legislation (Bateman *et al*., 2011).

Until the mid 1970's, the drainage and infilling of wetlands was a common and encouraged activity (Mitsch and Gosselink, 2007). Although people and governments have come to realise the importance of protecting these ecosystems with the Ramsar Treaty encouraging countries to take action on the matter, wetland loss continues to occur (Balmford *et al*., 2002). In many areas (such as New Zealand, Iowa and California) there have been up to 90% losses of wetlands over the past century as a result of anthropogenic activities (Mitsch and Gosselink, 2007; Tiner, 1984). It has been estimated that Europe has lost approximately two-thirds of its wetlands over the 20th century (EC, 1995), while the United States has lost approximately 54% of its total wetlands, with agricultural reclamation being the main driving force behind loss, followed by urbanisation (Chen *et al*., 2012; Tiner, 1984). This is likely to be due to the low perceived economic value of wetlands by landowners. Draining, dredging and infilling are the main methods responsible for this loss (Mitch and Gosselink, 2007). One useful method for assessing the impact that this infilling activity is having on the biota of sites is through the use of bioindicators and biomonitors.

1.4. Bioindicators and Biomonitors

A bioindicator can be described as a species or group of species the presence or absence of which reflects: the state of the environment (biotic or abiotic); impacts (from habitat to ecosystem level) of an environmental change; and the diversity within an area (McGeoch, 1998). Selection of suitable groups is based on a list of criteria described by Speight (1986) and McGeoch (1998). Groups which have been shown to be useful as indicators of habitat change include plants (LaPaix *et*

al., 2009; Pardo *et al*., 2011) and flies (Insecta: Diptera) at family and morphospecies levels (King & Brazner, 1999; Hughes *et al*., 2000; Whiles and Goldowitz, 2001), and in particular sciomyzids (Diptera: Sciomyzidae), commonly called snail-killing flies for wetlands (Williams *et al*., 2007; Williams *et al*., 2009; Murphy *et al*., 2012). The composition of plant communities has been shown to reflect their environmental conditions such as soil nutrient availability (Dickson and Gross, 2013), hydrology (Nishimoto and Hada, 2013) and soil disturbance (Marcelino *et al*., 2013; Nishimoto and Hada, 2013), making them suitable for assessing the impacts of wetland loss through infilling with C&D waste.

Diptera are a very large and diverse group of animals, with over 160,000 known species, and many more likely to exist (estimated that they comprise 15 - 20 % of all animal species; Marshall, 2012). They are found in a wide variety of habitats and provide many important ecological functions as a group, including biological control, plant pollination, and breaking down dung and carrion (Marshall, 2012). Dipteran communities are known to be sensitive to environmental conditions such as vegetation structure and moisture (Hughes *et al*., 2000; King & Brazner, 1999; Ryder *et al*., 2005; Whiles and Goldowitz, 2001). Identification of individuals can easily be made to family level using several keys (Unwin 1981; Oosterbroek 2006). Using taxonomic minimalism, groups with different morphological features can be identified within these dipteran families and treated as separate morphospecies (Beattie and Oliver 1994). This has the major benefit of allowing a high number of samples to be analysed (Beattie and Oliver 1994) and although individual species cannot be identified, it is still a useful method to assess biodiversity (Rivers-Moore and Samways 1996). Although many families of higher Diptera are commonly found in freshwater wetlands, among the most widely studied and useful as bioindicators are Sciomyzidae (Keiper *et al*., 2002).

Sciomyzidae are a large dipteran family that can be found in a wide variety of habitats from ponds to woodlands (Knutson and Vala, 2011; Williams *et al*., 2009). The life cycle of sciomyzids generally involves an aquatic or semi-aquatic larval stage that feeds on a variety of molluscs (Knutson and Vala, 2011) almost without exception (Vala *et al*., 2000). It is this mollusc-eating trait that also allows sciomyzids to have the potential to act as important bio-control agents for

pestiferous slugs and snails (Choi *et al*., 2004; Hynes *et al*., 2014; McDonnell *et al*., 2014). Collections can easily be made using a variety of methods including sweep nets, pan traps, malaise traps and emergence traps (Knutson and Vala, 2011; Williams *et al*., 2009) and identifications can be made to species level using keys by Rozkošný (1984 and 1987). This, combined with the low dispersal habits and habitat specificity of species (Knutson and Vala, 2011; Williams *et al*., 2007; Vala & Brunel, 1987; Speight, 2004), means that sciomyzids have been used successfully as bioindicators of environmental (particularly hydrological and vegetation structure) conditions in the past on wetlands (Murphy *et al*., 2012; Speight, 1986; Williams *et al*., 2009; Williams *et al*., 2010). Bioindicators, which provide qualitative information on the environment (Phillips and Rainbow, 1993; Markert *et al*., 2003), are not suitable to give quantitative information on any potential metal contamination from C&D waste.

Terms such as ‘heavy metals’, ‘trace metals’ and ‘trace elements’ have been used to refer to the metal and metalloid elements which are usually found at low background concentrations in the environment (Alloway, 2013; Fay *et al*., 2007). Many of these metals are essential for organisms to function correctly, but their presence in elevated concentrations (which varies for each metal and organism) can be toxic (Alloway, 2013). Although metals are found in all natural environments and soils around the world, anthropogenic activities such as mining, waste disposal, agriculture and fossil fuel combustion have resulted in areas with modified metal biogeochemical cycles and elevated environmental concentrations of metals (Alloway, 2013). Among the most environmentally important of these are arsenic, cadmium, chromium, cobalt, copper, lead, manganese, nickel, selenium and zinc (Alloway, 2013). Although C&D waste should be inert, it is often found that the waste contains some hazardous (in terms of metals and other contaminants) material, mainly as a result of poor sorting (Roussat *et al*., 2008). Six of the above most environmentally important metals (arsenic, cadmium, chromium, copper, lead and manganese; Alloway, 2013) have been found in the leachate of C&D waste at significantly elevated concentrations, as have aluminium, iron and other contaminants (fluoride, sulphate, dissolved organic carbon, polycyclic aromatic hydrocarbons and total dissolved solids) (Melendez, 1996; Roussat *et al*., 2008; Torgal and Jalali, 2011; Weber *et al*., 2002).

Environmental metal contamination can be measured in several ways, including direct sediment analysis (Alloway, 2013) and water analysis (Henry, 2014). Although these methods do have some advantages including the exact determination of abiotic environmental contamination at a particular place and time, they do have some shortcomings. It may be difficult to ensure that the samples analysed are representative of the surrounding area. These methods also do not account for temporal variation in concentrations or the bioavilability of the determined elements, as total soil and water metal concentrations may not reflect the potential toxicity of a site. Many of these shortcomings can be addressed through the use of biomonitors. A biomonitor of metals is an organism whose tissues accumulate metals, the concentration of which can give quantitative information on environmental background concentrations (Phillips and Rainbow, 1993; Markert *et al*., 2003) without many of the above mentioned problems associated with direct analysis of leachate and C&D waste.

It has been shown that Gastropods (Mollusca) are among the most effective groups of invertebrates for accumulating metals in their tissues (Dallinger *et al*. 2001; Salánki *et al*. 2003). Gastropods include snails and slugs, and are broadly described as having a distinct head that has tentacles and eyes, a flattened foot and a mantle covering (at least partly) the dorsal visceral mass (South, 1992). They are generally considered to be a suitable biomonitor group for metals and satisfy the criteria for choosing a suitable biomonitor (Triebskorn and Köhler, 1996; Dallinger *et al*. 2001; Salánki *et al*. 2003). Gastropods have a geographically widespread distribution, giving an international relevance to studies, and as slugs and snails, they have obvious limited mobility (South, 1992; Wiktor, 2000). Many of their life cycles are well understood, they are easily collected and many species can be easily identified (South, 1992; Wiktor, 2000).

Gastropods can accumulate metals through ingested food/soil and through adsorption directly from their surroundings (Gomot-de Vaufleury 2009; Croteau and Luoma 2008; Laskowski and Hopkin 1996; Notten *et al*. 2005). Notten *et al*. (2005) found that for the terrestrial snail *Cepaea nemoralis*, indirect metal (Zn, Cu, Cd and Pb) uptake through plants (*Urtica dioica*) is the most important and effective uptake method, however there was still found to be some uptake directly from the soil. This direct uptake was thought to have resulted from both ingested

soil particles and contact of the body with the soil beneath (Notten *et al*. 2005). It is known that many terrestrial molluscs ingest soil, which may possibly provide additional nutrition (Gomot *et al*. 1989). Free metal ions are the most easily bioaccumulated form of metal (Spurgeon *et al*., 2006; Hough *et al*., 2005). Competition is known to occur between different free ions (both metals and H+) for the cell membrane uptake sites (Spurgeon *et al*., 2006). The H+ ions (more plentiful in lower pH soils), along with Ca+ and Mg+, can block these sorption sites, so reducing the uptake of metals in the tissue (Spurgeon *et al*., 2006). On the other hand, some metals can have synergistic effects on each other's uptake (Peijenburg, 2002), though ultimately bioaccumulation rates are specific to the species of both metal and organism.

1.5. Scope and objectives of this study

There has been some work carried out on the broad-scale amount of C&D waste disposed of (Schrör, 2011; Symonds Group Ltd. *et al*., 1999; USEPA, 2009), and its potential contaminative impacts on the environment (Melendez, 1996; Roussat *et al*., 2008; Torgal and Jalali, 2011; Weber *et al*., 2002). There is, however, a lack of knowledge regarding the spatio-temporal distribution of those infill sites at a local scale, the habitat types being lost to infill, the ecological implications of using C&D waste to infill wetlands, and the potential bioavailability of metals within the waste. Just one study by Gabrey (1997) observed the impact of C&D waste on bird populations (regarding implications for air traffic at airports; no significant impact was found), and no studies (to the author's knowledge) have looked at any other taxonomic group.

As these infill sites are so common around the world, it is important to gain an understanding of which habitats types are being lost and what impact, if any, they have on the biota and ecological communities therein. This provided the incentive for this study using County Galway, Ireland as a case study area, the objectives of which are to:

1. Assess the spatio-temporal distribution patterns of C&D waste infill sites, both legal and illegal, at a local scale and identify issues with non-compliance on legal sites.

2. Assess the qualitative and quantitative impacts of infilling wetlands with C&D waste on soil, and plant and dipteran communities.

3. Employ the Grey Field Slug (*Deroceras reticulatum*) as the first biomonitor of metals in construction and demolition waste used to infill wetlands.

1.6. Structure of this thesis

This thesis follows a papers-based format and consists of three papers.

The first paper looks at the spatio-temporal distribution patterns of C&D waste infill sites and non-compliance issues using a case study of a single county in Ireland. The second paper investigates the ecological impact that results from infilling wetlands with C&D waste. The third paper investigates the bioavailability of metals on C&D waste infill sites using a biomonitor species (*Deroceras reticulatum*).

All three studies use C&D waste sites within County Galway, with the third paper using additional comparative sites (mines) in County Tipperary.

Due to this thesis format, there is some necessary repetition between the separate papers.

1.7. References

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Chapter 2:

Spatio-temporal distribution of construction and demolition (C&D) waste disposal on wetlands: a case study

This paper has been submitted to *Land Use Policy* for consideration for publication.

2. Spatio-temporal distribution of construction and demolition (C&D) waste disposal on wetlands: a case study

2.1. Abstract

Although infilling of wetlands (legal and illegal) is commonplace, little is known about the spatio-temporal distribution of construction & demolition (C&D) waste infill sites at a local scale. This is particularly important given the multiple functions of wetlands including, *inter alia*, habitat provision, flood control and water storage. This case study quantifies, for the first time, the use of wetland habitats for C&D waste infilling at a local scale in addition to identifying patterns of C&D waste site distribution and recording issues of non-compliance. We found that wet grasslands and peatlands were the most commonly infilled habitats, particularly near urban areas and adjacent to major roads. Of greater concern was that over 40 % of C&D waste sites granted permits were within 1km from Special Areas of Conservation (EU Habitats Directive) and 54% were located on aquifers of extreme vulnerability. We found that the conditions attached to infilling permits were frequently broken and commonly occurring illegal infilling sites had similar distribution patterns to the legal sites. Providing local authorities with sufficient resources to effectively police these sites in combination with examining alternative uses for C&D waste e.g. recycling, are likely to be the most effective ways of dealing with these issues. More rigorous ecological investigations of proposed infilling sites prior to granting of permits, would also limit the number of wetlands affected by infilling.

2.2. Introduction

Construction and demolition waste is produced during the construction, renovation and demolition of structures (USEPA, 2009). The constituents of C&D waste can vary regionally, but include concrete, asphalt/bitumen, timber, gypsum, metals, plastics, ceramics, glass, soil and stones (Symonds Group Ltd., 1999; Poon *et al*., 2001; Franklin Associates, 1999; Hyder Consulting *et al*., 2011). In the European Waste Catalogue and Hazardous Waste List (Environmental Protection Agency (EPA), 2002), 16 out of a total of 28 sub-categories of C&D waste are classified as hazardous. Leachate generated from C&D waste can also contain significantly elevated levels of metal contaminants including Al, As, Cd, Cu, Fe, Pb and Mn (Melendez, 1996; Weber *et al*., 2002; Torgal and Jalali, 2011), two (As, Cd) of which are EU (Council Directive, 2008/105/EC) and USEPA (USEPA, 2013) priority pollutants. In addition, recent research shows that terrestrial slugs collected on C&D waste have significantly elevated concentrations of As, Ba, Cd, Co, Sb, Se and Tl, compared to slugs collected on control sites (See Chapter 4).

The most recent data indicates that the United States produced an estimated 170 million tonnes of building-related C&D waste in 2003, excluding C&D waste from other sources such as roads and natural disasters (USEPA, 2009). While there were approximately 1,900 C&D waste specific landfills in the US during the mid-1990s (no known surveys have been carried out since), undocumented infilling was common (Franklin Associates, 1998; USEPA, 2014) with the state of Georgia alone reporting 900 unpermitted sites (ICF, 1995). In the EU, C&D waste accounted for 32.9 % (859 million tonnes) of total waste production (the largest waste component in the EU) in 2008, with France being the largest contributor, at almost 253 million tonnes (Schrör, 2011). Production of C&D waste in the Republic of Ireland (Fig 2.1) which peaked at 17.8 million tonnes in 2007 (EPA, 2009a) was followed by production levels of just over three million tonnes in 2011 (EPA 2013). This drop in C&D waste production is the result of the recent economic recession (beginning in 2007/2008) which led to a fall in house prices, and, therefore, a reduction in construction sector activities (ESRI, 2014).

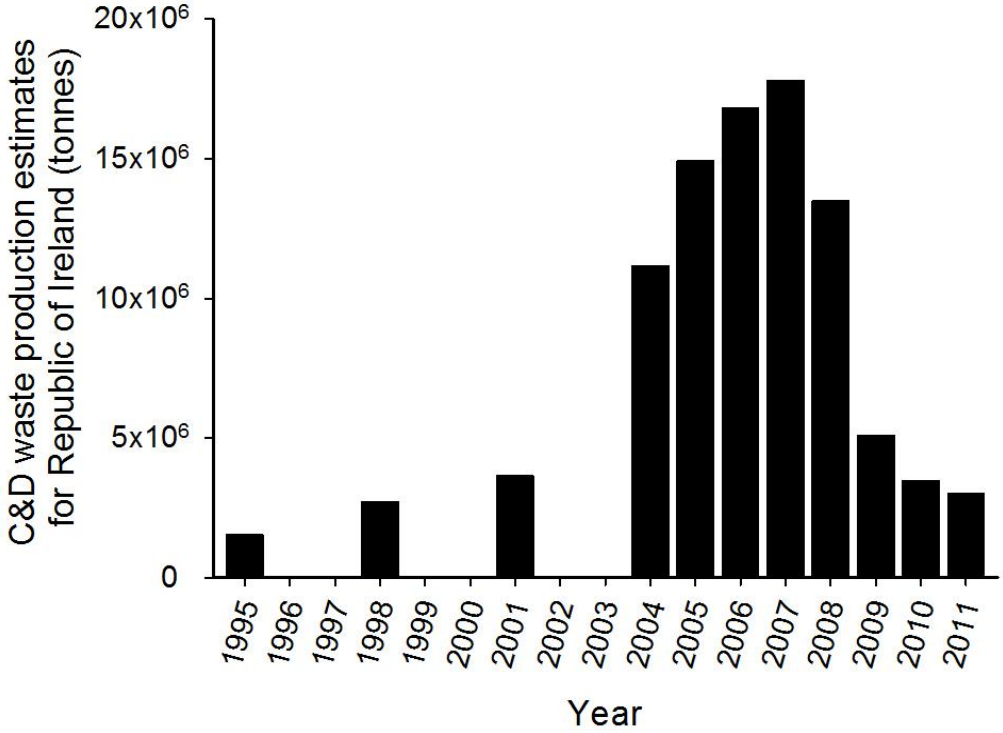

Fig 2.1. *Construction and demolition (C&D) waste production estimates for Ireland (1995 - 2011; EPA, 1996; 1998; 2001; 2004; 2006; 2008; 2009a; 2009b; 2011; 2012; 2013).*

Despite the many essential ecosystem services provided by wetlands e.g. habitat provision, flood control, water storage and recreation (Costanza *et al*., 1997; Keddy, 2000; Lehner and Döll, 2004, Millennium Assessment Report, 2005), wetland reclamation for conversion to improved agricultural land by infilling with C&D waste is commonplace globally (Poon, 2001; Shen *et al*., 2004; Mitsch and Gosselink, 2007; Chen *et al*., 2012). This leads to habitat destruction resulting in negative impacts on the associated biota (Staunton et. al., 2014; See Chapter 3). In addition, there is increased potential for environmental contamination from harmful substances contained within the waste (See Chapter 4).

A summary of the legislative framework for C&D waste disposal in the Republic of Ireland is given in Fig 2.2. While the Waste Management Act (Statutory Instrument No. 10/1996) was enacted in Ireland in 1996 to give effect to preceding EU waste and habitats directives, there was no separate regulation for C&D waste disposal until 2001, largely because it was perceived as being inert

waste (Symonds Group Ltd., 1999; Local Authority staff member, Personal communication). Although all C&D waste should have been disposed of within municipal landfills, it often ended up being used as unpermitted fill material either on site (of source) or elsewhere in the locality (Symonds Group Ltd., 1999; Local Authority staff member, Personal communication). Between 2001 and 2008, Irish local authorities granted Waste Permits (WP) specifically for sites receiving < 5,000 tonnes of C&D waste per annum (Statutory Instrument No. 165/1998; RPS-MCOS, 2004; Fig 2.2). For sites receiving > 5,000 tonnes per annum, a waste license had to be obtained from the Environmental Protection Agency (EPA), a much more rigorous process which required completion of Environmental Impact Assessments (EIA) and had longer application processing times (RPS-MCOS, 2004).

The application system was changed with the passing of the 2008 European Directive on waste (Council Directive 2008/98/EC) and the Waste Management (Facility Permit and Registration) Regulations, 2007 (S.I. No. 821 of 2007). Since June 2008, EPA standardised applications (Office of Environmental Enforcement, 2008) are made to local authorities for either a Certificate of Registration (COR; to accept ≤10,000 tonnes of C&D waste per annum) or a Waste Facility Permit (WFP; to accept 10,000 - 50,000 tonnes per annum; Fig 2.2). Waste licenses, issued by the EPA, are required for sites that propose to accept more than 50,000 tonnes per annum (or for waste containing > 15% residual waste for disposal).

In Ireland, all permits (i.e. WP, COR and WFPs) are issued with compulsory conditions attached for the permit holders and landowners (Statutory Instrument No. 165/1998; Office of Environmental Enforcement, 2008). These conditions may vary a little according to site-specific parameters such as location (EPA staff member, Personal communication) but there is some commonality, i.e. that the site is secured to prevent trespassing; that waste (volume and content) is as permitted; that environmental safeguards and mitigation measures are implemented; and that the site is finished adequately after infilling (this usually involves waste levelling, application of top soil and reseeding for agriculture).

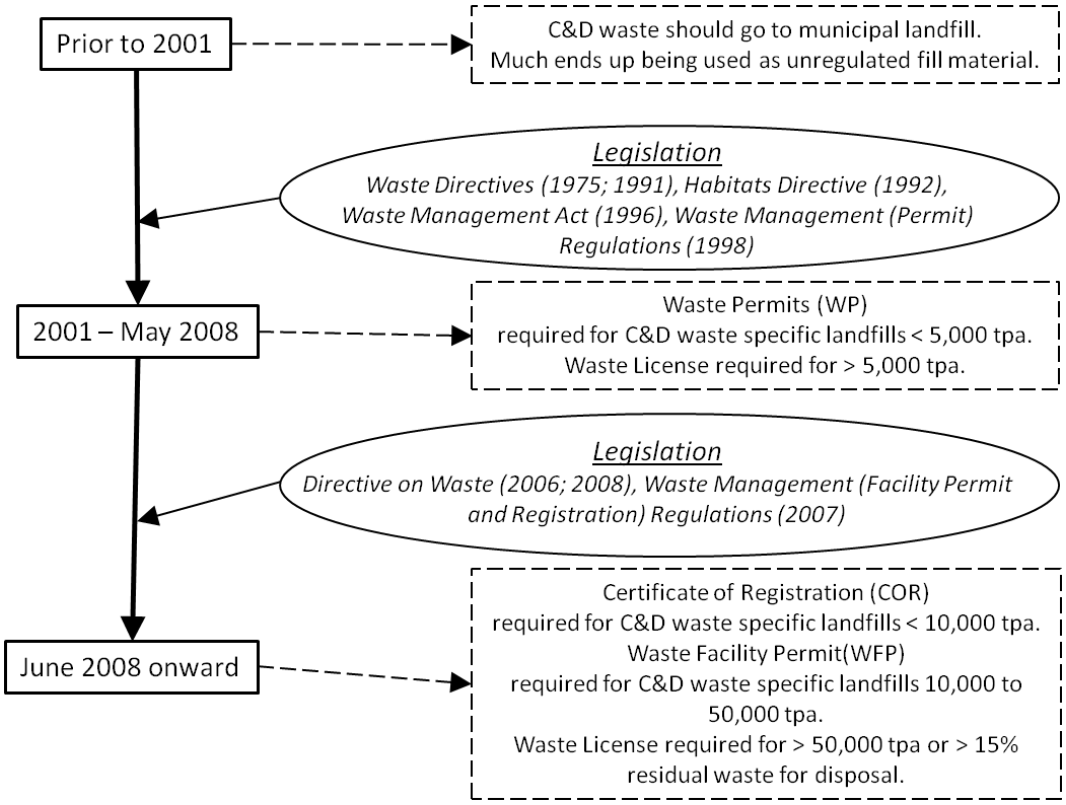

***Fig 2.2**. Brief summary of the legislative history of C&D waste disposal in the Republic of Ireland.*

While reuse, recycling and recovery are preferred over disposal (Peng *et al*., 1997, Council Directive 2006/12/EC, Council Directive 2008/98/EC), best practice is the prevention and minimisation of C&D waste at source (Poon *et al*., 2004; Tam and Tam, 2008; Teo and Loosemore, 2001). While this policy of waste minimisation has been favoured by governing and advisory bodies such as the USEPA, the Irish EPA and Irish National Construction and Demolition Waste Council (USEPA, 2014; NCDWC, 2006), the infilling of wetlands has been and continues to be commonplace practice in many countries. There is currently a paucity of quantitative data regarding how effectively the legislation is implemented on the ground or indeed how lands (particularly wetlands) are utilised for the disposal of C&D waste. This provided the incentive for this study where:

- spatio-temporal distribution patterns of C&D waste infill sites with reference to wetlands are quantified, for the first time, using a local authority case study
- levels of non-compliance within the local authority area in relation to land coverage of waste, waste types and unregulated disposal are established
- barriers to effective C&D waste recycling in the local authority area are discussed in the context of providing recommendations for future policy makers and regulators to reduce the environmental impact of C&D waste.

2.3. Methods

2.3.1. Study area

This study was based on available registered waste permit applications from an Irish local authority (6,148km^2) for the reclamation of land using C&D waste between 2001 and 2012. The county (and associated local authority), landowners and permit holders will remain anonymous for the purposes of this study. The data-set included all permitted sites, all sites where permission had been refused and all sites pending a decision (as of June, 2013). Any available information on site location, site area, ecology, site boundaries, permitted waste volume and constituents, permit status, infilling period and non-compliance was obtained from the waste permit application files held by the local authority. Some limited information (from 2008 - 2012) regarding 10 sites granted with COR (< 10,000 tpa) was also collected from an online EPA national database (EPA, 2014). No waste licenses (> 5,000 tpa) were granted specifically for C&D waste infilling in the county during the study period (2001 - 2012). In addition, 70 of the sites were visited at least once between 2009 and 2011 as part of related studies (Staunton *et al*., 2014a; 2014b; See Chapters 3 and 4) during which time observations regarding non-permitted activities such as fly-tipping, waste contamination, etc. were recorded. ArcMap 10 (ESRI), Microsoft Excel 2007 (Microsoft Corporation) and SPSS 20 (IBM Corporation) were used to analyse the data.

2.3.2. Mapping

Of the 167 application files initially identified, 133 files were made available and were used in this study. Geographical Information Systems (GIS) software (ArcMap 10) was used to map the location of the 133 sites throughout the county. For the 34 sites where files were not available, the townland addresses were known making it possible to map their approximate location and permit status, allowing for a better overview of the total site distribution (and permit status). Existing maps for habitat type, geology, soil type, aquifer classification and vulnerability, Natura 2000 designated areas, roads and urban areas were overlaid to contextualise the sites. Ortho-referenced aerial photography from 1995, 2000, 2005 and 2010 was also overlaid at each site to observe temporal changes in C&D waste coverage (OSI, 1995, 2000, 2005, 2010).

The most up-to-date aerial photographs (OSI, 2010) covering areas within five kilometres of the city and five largest towns in the county were also systematically examined, knowing that C&D waste is typically transported over short distances (< 25 km) for disposal (Reid, 2003). The number of areas on those photographs larger than 0.2 hectares that appeared to be infilled (addition of grey, C&D waste-like surface; site visits were not practical to all sites due to time constraints) for which there were no permits or permit applications were also noted.

2.4. Results and discussion

2.4.1. Spatial distribution of C&D waste infill sites

Even prior to the recent economic boom (ESRI, 2014) and its associated high production rates for C&D waste (Fig 2.1), the total wetland coverage in the local authority area is known to have decreased (through various means) from 33% of the total land area in 1990 to 31 % in 2000 (EPA 1990, 2000). With more recent data not currently available, the amount of wetlands lost to infilling and other activities since 2000 is unknown. Of the 167 applications made to the local authority for permits to use C&D waste as infill, 58.7 % were granted, 15% refused, 19.8 % pending a decision (as of 6th August 2013) with the remainder being invalid or withdrawn applications (Fig 2.3; Fig 2.4). Based solely on the

133 files viewed, 63.2 % were granted, 18 % refused and 18.8 % were still pending a decision (also as of 6th August 2013). Reasons for refusal, where available, included the creation of traffic hazards, flooding potential, incomplete applications, requirement of waste license, landscape modification, location on or adjacent to an SAC, noise pollution, and risk to waters and ecology (details were not available).

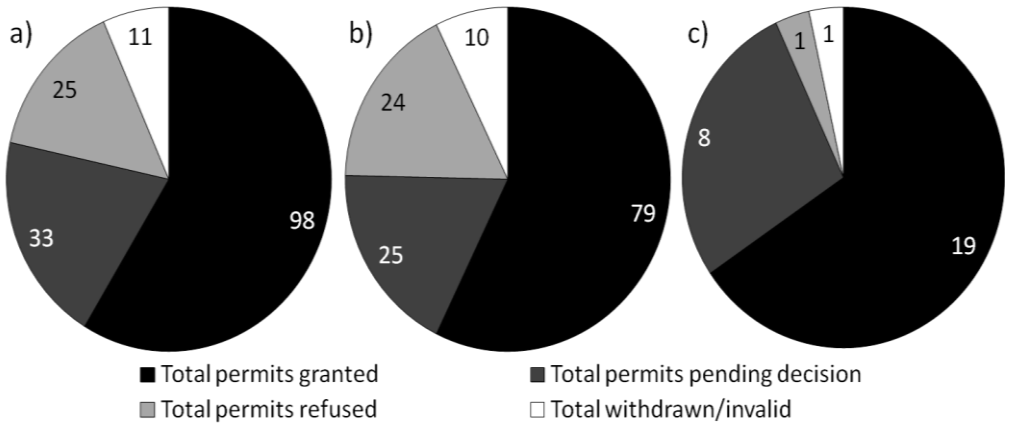

***Fig 2.3**. Status for C&D waste infill permit applications (n = 167) made to local authority in: a) Total 2001-2012; b) 2001-May 2008 (Waste Permits); c) June 2008-2012 (Certificate of Registration and Waste Facility Permit)*

The original site habitats prior to infilling (only sites which were granted permits for which files were available; n = 84) varied from peatland to wet grassland (Table 2.1). This assessment was based on information extracted from a combination of permit application documents (only rarely stated and with little detail), along with GIS habitat maps, aerial photography and site visits by the senior author using Fossit (2000). The majority (57 %; n = 48 of 84) of sites were located on wet grassland (of varying management intensities from intensive to extensive), with a permitted infilling area covering a total of almost 170 ha, followed by peatlands (23 %; n = 19 of 84) covering a total 45 ha. The favouring of wet grassland over other wetland types is likely to be due to the high coverage of grasslands in the eastern half of the county (EPA, 2000) where permit density is highest (Fig 2.4), as is population density (Central Statistics Office, 2011). The western half of the county is dominated by peatland habitats (EPA, 2000). A chi-squared test shows that the frequency of habitat type infilled across all sites is

significantly ($P < 0.01$; $X^2 = 206.49$) non-random, confirming that preferential infilling of certain habitat types (i.e. wet grasslands) has occurred. There was no significant relationship found between the granting or refusal of permits and the habitat type (OSI, 2010; Table 2.1). Permit application files only rarely mentioned possible ecological impacts or suitable mitigation measures, with these being minimal and vague.

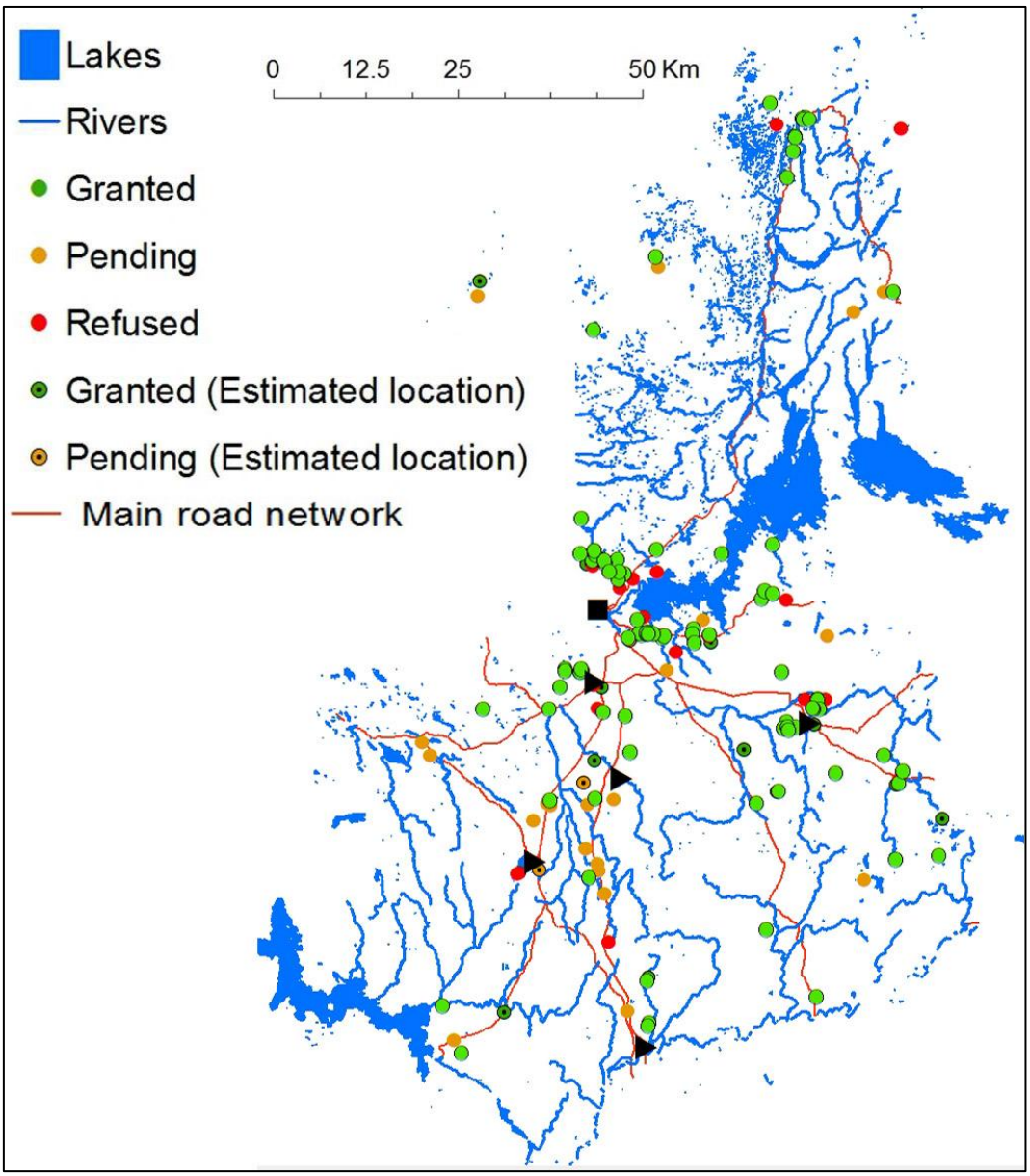

***Fig 2.4**. Distribution of C&D waste application sites throughout the county in Ireland, showing county city and towns. Sites with estimated location are shown at the townland scale.*

***Table 2.1**. Details of habitat types which were granted permits (n = 84) for infilling with C&D waste in local authority area, Ireland. 'Grassland' categories include pastures farmed at varying intensities. These data include only sites that had files available. Habitats were determined using a combination of on-file information, GIS habitat maps, site visits (Fossit, 2000) and aerial photography.*

Habitat (dominant) types infilled on granted sites	Sites granted permits (% of total granted sites; n = 84)	Total area permitted for infill on sites (ha)	Sites containing infill in 2010 (% of those with permits)	Total area infilled based on 2010 aerial photography (% of permitted area)
Wet grassland	57.1	169.132	87.5	67.6
Peatland	22.6	45.114	100	85.6
Wet grassland/peatland mosaic	8.3	20.013	62.5	16.6
Wet grassland/dry grassland mosaic	6	7.837	100	119.6
Quarry*	2.4	12.735	50	10.4
Dry grassland**	1.2	0.247	0	-
Dry grassland/reed and large sedge swamp mosaic	1.2	2.86	100	32.9
Reed and large sedge swamp/peatland mosaic	1.2	2.35	100	7.3

*Determining the area filled with C&D waste in quarries is extremely difficult via aerial photography, and this is likely to be an underestimation.
**Dry grassland site had no infill present.

In the United States, legislation prevents the infilling of wetlands where possible (Wisconsin Department of Natural Resources, 2014). When a permit application is made in the US for infilling a wetland, the applicant must show that there are no other options (different locations, waste treatment methods, etc.) available, to ensure the option with the least environmental impact will be chosen for C&D waste disposal. If an area of wetland is to be infilled, some mitigation measures (such as earmarking an area for conservation) must also be taken by the applicant

to ensure the protection of the remaining wetland area (Wisconsin Department of Natural Resources, 2014). Such site selection and mitigation strategies should be adopted internationally to promote the preservation of the associated ecosystem services.

Using GIS analysis, it was found that four sites (out of n = 84) are partially on, or immediately adjacent to a Special Area of Conservation (SAC; EU designation), and a further four immediately adjacent to a Natural Heritage Area (NHA; Irish designation). A total of 37 (44 %) C&D waste infill sites were granted within 1 km of an SAC, where a significant ($P < 0.01$; Pearson correlation; linear regression analysis) negative relationship was found between site density and increasing distance from the SAC (Fig 2.5). The cumulative impacts of infilling wetlands adjacent to SACs is likely to pose future problems for those protected areas particularly in the context of contamination (Staunton *et al*., 2014a; See Chapter 4), ecological change (Staunton *et al*., 2014b; See Chapter 3) and water displacement caused by the waste. Environmental studies such as a full Appropriate Assessment (AA) should be a base requirement for all permit applications near protected wetlands. Presently AA screening is carried out by the local authority for sites within 15 km of the designated lands, with a full AA required only for those that are thought likely to impact the SAC (i.e. sites that screen in) (National Parks & Wildlife Service (NPWS), 2010; Local Authority staff member, Personal Communication). It was also found that 53.8 % of sites with granted permits were located on aquifers of extreme vulnerability (GSI, 2006), with 26.5 % of these being situated on karstic aquifers. This is of great concern, as research has shown that the C&D waste and its leachate can contain elevated concentrations of contaminants (Melendez, 1996; Weber *et al*., 2002; Torgal and Jalali, 2011), some of which are in bioavailable forms (See Chapter 4).

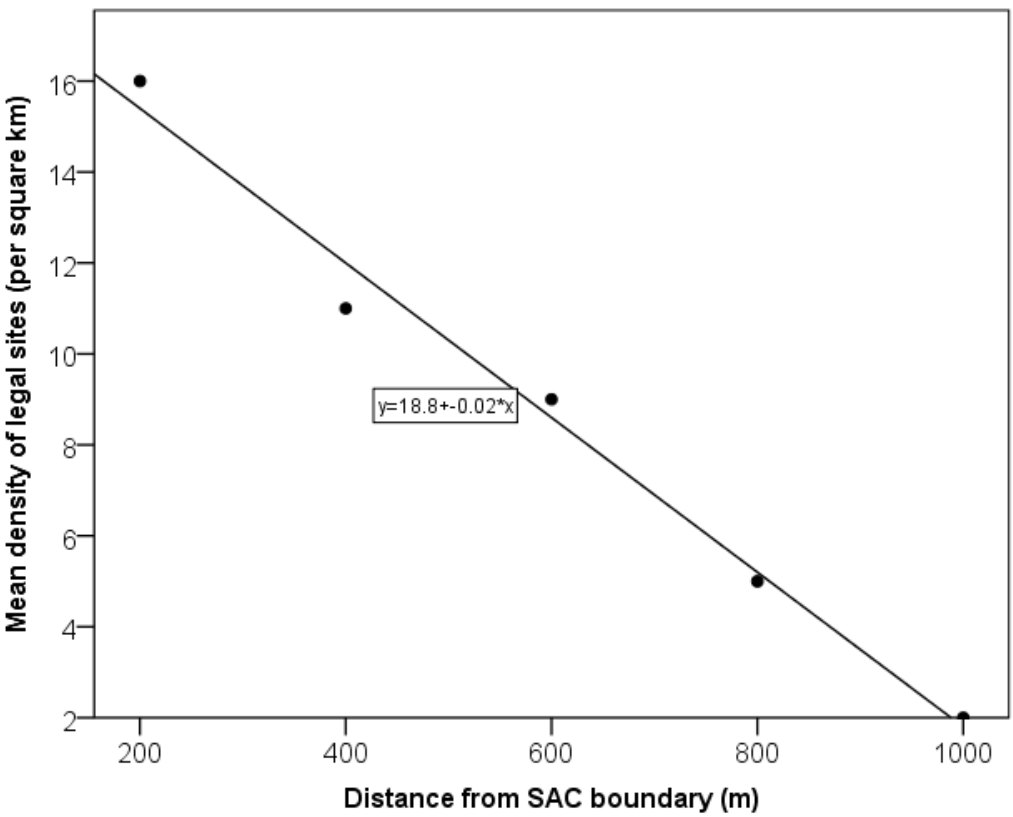

***Fig 2.5**. Scatterplot showing the number of C&D waste infill sites near Special Areas of Conservation (SAC) in increment distances of 200m for the study county, Ireland. Pearson correlation, linear regression analysis shows a significant negative correlation between the number of sites and the distance from an SAC (P = 0.001).*

Although the proportions of applications with either positive or negative outcomes (granted or refused) changed slightly for CORs/WFPs (June 2008 - 2012) when compared to WPs (2001 - May 2008), this change was not found to be significant (Chi Squared test; P = 0.19). The number of permits granted for each year from 2001 to 2012 can be seen in Fig 2.6. The recent economic recession (beginning in 2007/2008; ESRI, 2014) and associated reduction in C&D waste production (Fig 2.1) is most likely to be the main cause of the reduced number of permits post 2008 (Figs 2.3 and 2.5). A more detailed application process from June 2008 may also have dissuaded those who were less likely to be granted a permit. Wetland loss through infilling is, therefore, likely to increase during times of economic growth. Extensive pipe and cable laying works ran from May 2010 until August 2012 in Town 5, producing large amounts of C&D waste (Local Authority staff

member, Personal communication) with seven CORs being granted in that time within 10km of the town centre. This is likely to have contributed to the temporary increase in permits granted during 2010 (Fig 2.6).

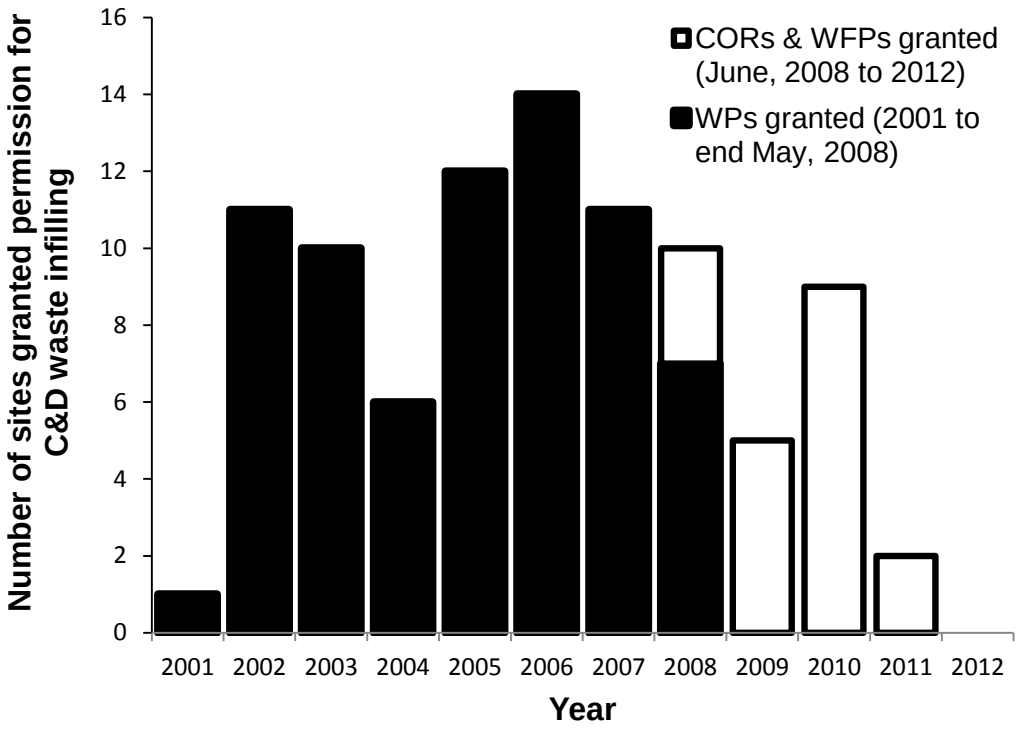

***Fig 2.6**. Temporal variation in annual number of granted C&D waste infill permits from 2001-2012 by local authority, Ireland. This only includes granted permits with available files (total n = 91); WP = Waste Permit; COR = Certificate of Registration; WFP = Waste Facility Permit.*

The spatial distribution of C&D sites is focused around urban areas (in particular the city) and the main road network (Fig 2.4). Significant spatial clustering of sites was confirmed with cluster analysis using GIS software (Fig 2.7). Site density was found to be significantly negatively associated with increasing distance (up to 10 km) from the city boundary (Linear regression analysis, Pearson correlation; r = -0.737; $P < 0.01$; this distance was attributable to 60 % of site density) and from all major urban areas (Linear regression analysis, Pearson correlation; r = -0.774; $P < 0.01$; this distance was attributable to 70.1% of site density) (Figs. 2.8 and 2.9). This is reflected in the density of granted permits

shown in Fig 2.10, with increasing distance from the urban areas, particularly the county city, where a total of 31 applications (2001-2012) were granted permits within 5 km of city limits. This high density of C&D waste sites adjacent to urban areas is understandable, as a short transport distance will have clear cost savings. This may mean that unprotected wetlands located near large urban areas may come under increased pressure compared to those further away. Such a situation would highlight the need to ensure that infill sites are selected primarily based on the lowest environmental impact, with disposal convenience coming as a secondary selection criterion.

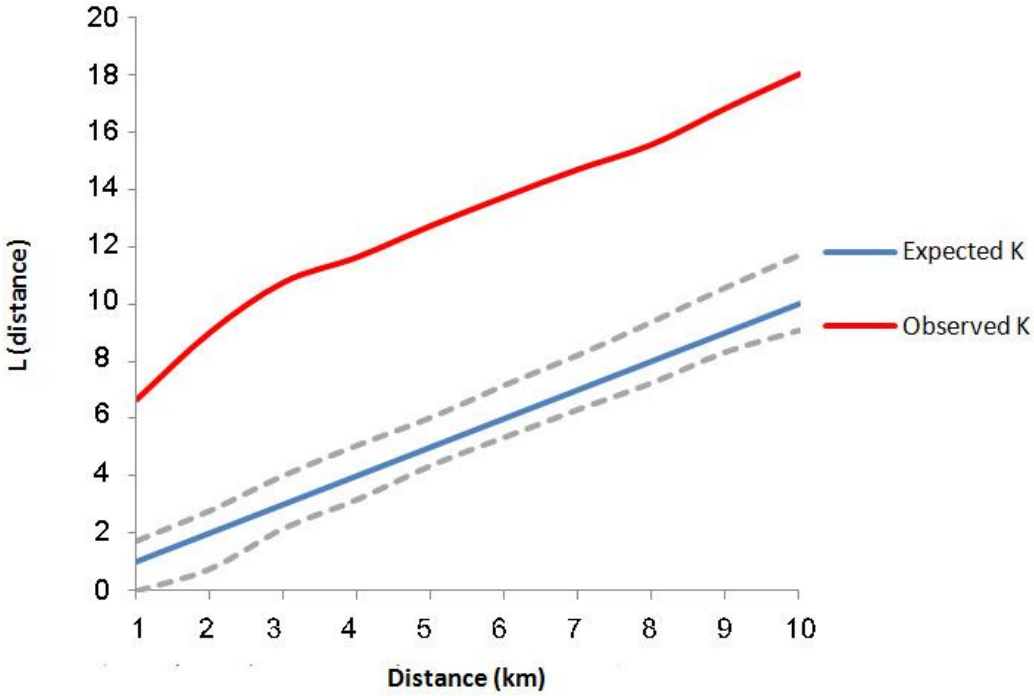

Fig 2.7*. Cluster analysis (Ripleys K function) for the distribution of C&D waste infill sites in the local authority, Ireland. X axis shows between site distance for analysis. Y axis shows Ripley's K function: L distance). If observed K is greater than 95 % confidence envelope (dashed line) then sites are significantly clustered.*

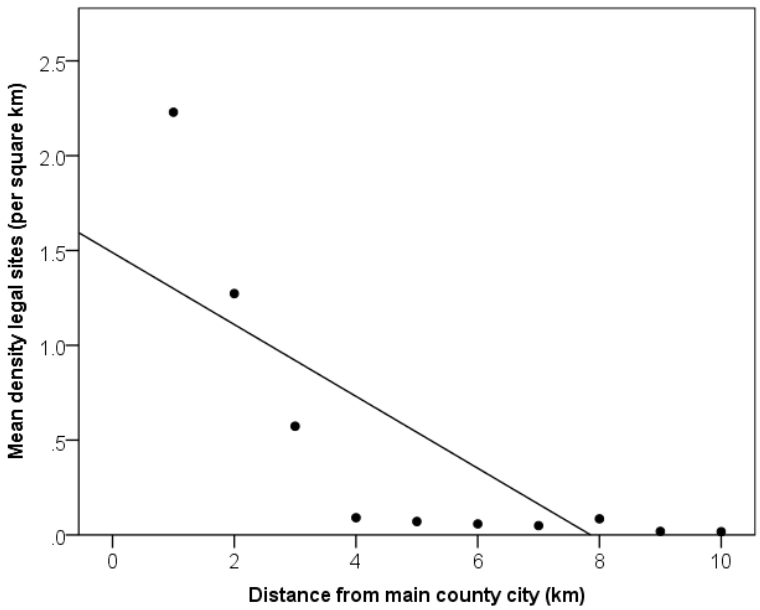

Fig 2.8. *Scatterplot showing the mean density (per km^2) of C&D waste infill sites near the main city for the study county, Ireland. Linear regression analysis (Pearson) shows a significant negative correlation between the number of sites and the distance from an urban boundary. P < 0.01, r = -0.737.*

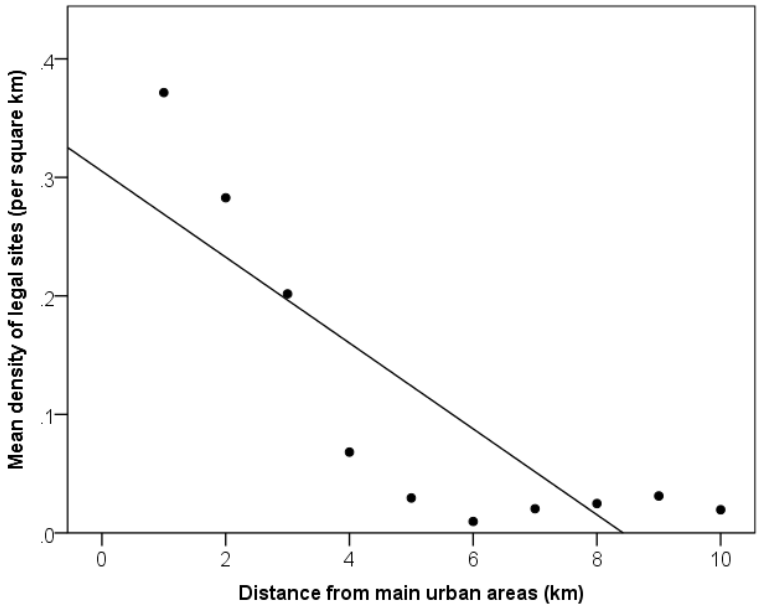

Fig 2.9. *Scatterplot showing the mean density (per km^2) of C&D waste infill sites near urban boundaries (for county city and five largest towns combined) for the study county, Ireland. Linear regression analysis (Pearson) shows a significant negative correlation between the number of sites and the distance from an urban boundary. P < 0.01, r = -0.774.*

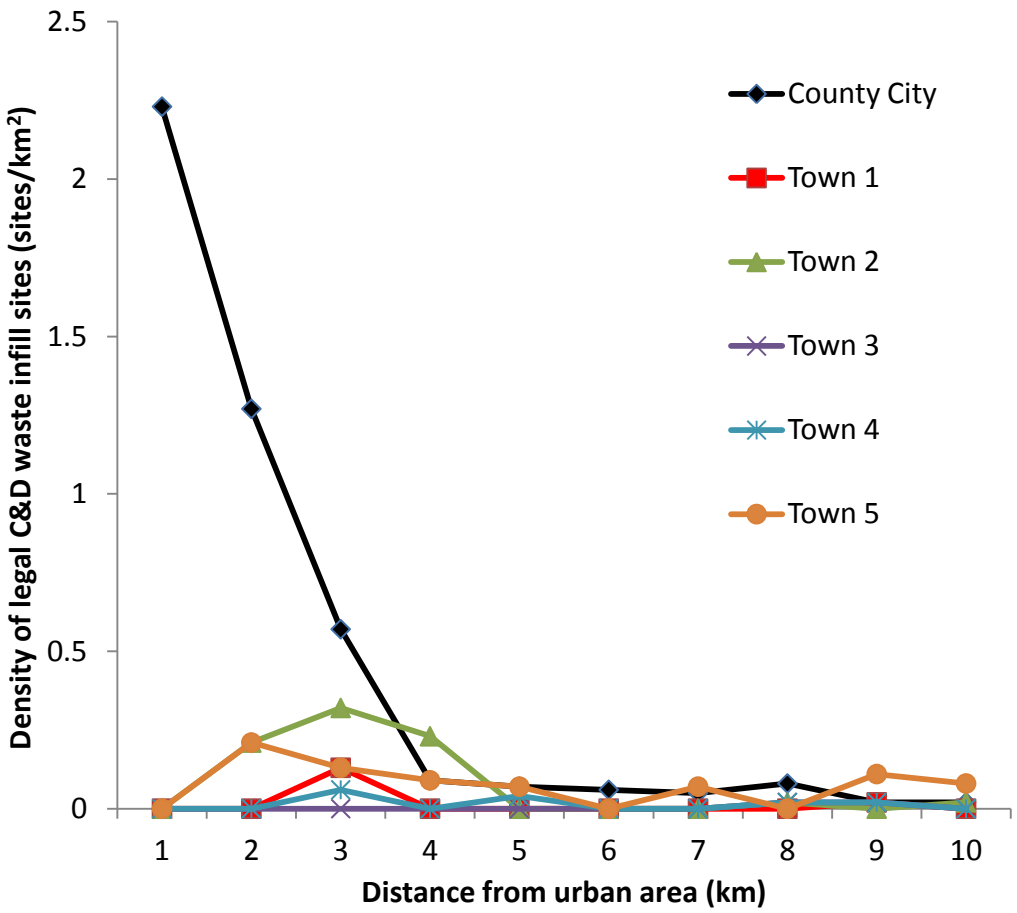

***Fig 2.10**. Density of granted permit (2001-2012) sites around the county city and the five most populated towns (population; Central Statistics Office, 2012) in the local authority case study, Ireland.*

Throughout the county, C&D waste sites were found to be most often located adjacent to the main road networks. Using the GIS outputs of the study, the distances from each infilled area to the nearest road (excluding tracks) were determined, with a median of 4.3m and mean of 28.1 m (± 66.1) for granted waste permits (indicating that most sites are directly adjacent to roads). These findings are understandable, as transporting the waste across fields with large machinery would be difficult in any location, especially in wet conditions. As the total road length can also be determined from GIS software, the tendency for these sites to be located within 200 m of main roads can also be portrayed as one site for every 21.1 km, 55km and 213.7 km for national roads, regional roads and third class roads respectively (Statutory Instrument No. 14/1993). These data indicate that preference for site location appears to be given to sites that are easily accessed by

major roads. While this is a clear advantage in terms of ease of waste transport with large vehicles, it does mean that wetlands adjacent to major roads have an increased threat from infilling than more isolated wetlands.

2.4.2. Problems identified with site monitoring

It was found that only 73 % of sites which had been granted permits and for which files were available (n = 84) actually contain infill. Out of 25 sites which were still pending a permit decision in June 2013, and 24 other sites which were refused permits, 52 % and 54 %, respectively, actually contained infill, based on aerial photography examination. The absence of infill in sites which have permits is likely to reflect the recent economic recession. However, the presence of infill on so many sites which were refused permits or had not yet been granted permits is an indication of the difficulty associated with monitoring such sites and enforcing the associated regulations. It also suggests that landowners do not regard the activity as potentially harmful, and have little understanding of the importance of the permitting system. Bringing legal proceedings against such landowners is perceived as being too expensive for the Local Authorities (Local Authority Staff Member, Personal communication).

Although information regarding the content of the C&D waste could not be found on some of the application documents, 88.8 % of files (granted and pending a decision) had a brief description (such as “concrete”, “brick”, “ceramic”, “soil and stone”) of the waste type or used the European Waste Catalogue codes (EPA, 2002). Soil and stone was listed on the majority of these files, with concrete, bricks and bituminous mixtures also being frequently listed (Fig 2.11). After visits to 70 sites between 2009 and 2012, it was found that mixing occurs to some extent between categories on the majority of sites, resulting in non-permissible items or substances being used as infill. Thirty sites (43 %) which were visited had clearly visible problems with illegal dumping or fly-tipping of some description (municipal, commercial or agricultural waste) on at least one occasion. Glass and/or timber was seen on 12 (17 %) sites, although no permission had been granted (according to files) for the inclusion of these materials. Access to all C&D waste infill sites should be restricted (particularly outside opening hours) under permit stipulations, but gates and boundary fences were frequently found to be

open, damaged or completely absent throughout the study, allowing easy access for fly-tipping of waste on the sites. Municipal, commercial and agricultural waste was seen partially buried on 18 (26 %) sites highlighting a lack of awareness among permit holders of the potential environmental problems associated with such activities. On one occasion the senior author witnessed agricultural waste being buried beneath the C&D waste at one site.

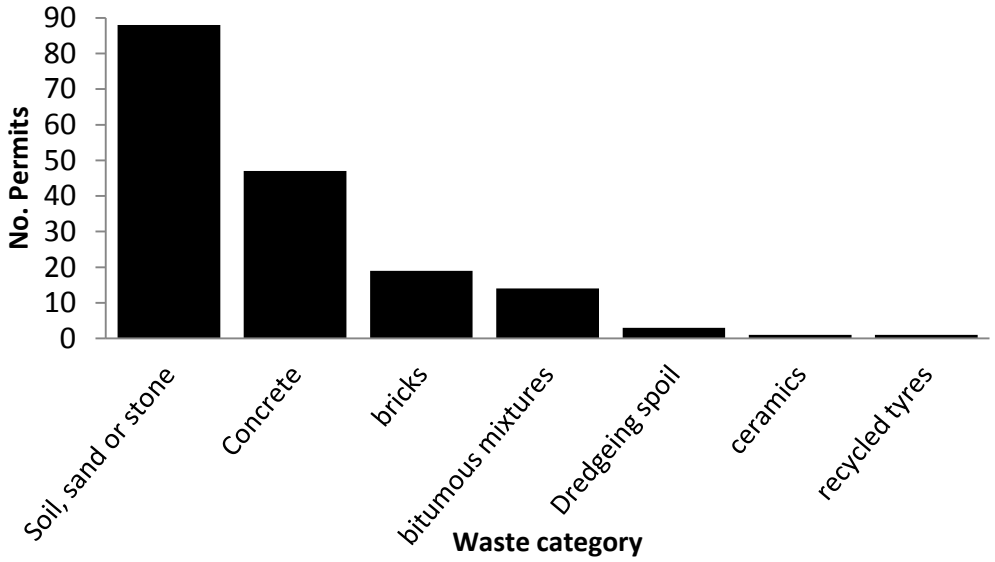

Fig 2.11. *C&D waste types listed on waste permit application files (n=133) in the study county, Ireland. Recycled tyres were used only as a surface over other C&D waste at one site as part of a planned sports facility development (it did not constitute the main infill material).*

Although official inspections are carried out periodically (usually annually; Local Authority staff member, Personal communication), a lack of resources means that a more effective and rigorous policing and inspection schedule cannot be carried out on these sites, and it is therefore difficult to enforce the various stipulations of the waste permits. Hiring constraints meant that during the course of this study, the Local Authority (covering approximately 6000 km^2) had just two staff members of whose many duties one was the completion of site inspections (Personal communication, Local Authority staff member). A description of exactly where the waste was expected to come from was only found in one (< 1 %) permit application (in that case it came from a commercial development in the county city). Some of the biggest and most 'active' sites, as decided on a case-by-

case basis, may have extra local authority site visits (Local Authority staff member, Personal communication).

Enforcing the limit regarding the amount of C&D waste these sites can accept is also a problematic task, given the number of sites, and the absence of on-site weighing facilities. This leaves it up to the permit holder to record the amount of waste being disposed of. One site, which had a permit to accept < 5,000 tonnes of C&D waste per year, had accepted an estimated 100,000-150,000 tonnes in the first year (2001; according to the application documents) of operation alone, with the C&D waste up to three metres deep (permitted to be up to a maximum of 0.5m according to the permit application file for that site). In that case the waste permit was terminated. Had the landowner initially applied for a waste license (as required for this waste volume), then an EIA would have been required, and the application processing time would have been longer (RPS-MCOS, 2004). Another similar difficulty is checking and enforcing the area limits covered by the C&D waste.

A total of 91 ha was permitted for infilling on all granted WPs (n = 79; i.e. only those using the 2001-2008 application system). However, based on the 2010 aerial photography (OSI, 2010) these sites had a total of 117 ha (129 % of permitted area) infilled, including a total of 55 ha infilled outside permit boundaries across 28 sites (> 0.25 ha per site). In addition, 29 ha of the lands within the permit boundaries had not been infilled (OSI, 2010). As these photographs were taken in late 2009/early 2010, some further infilling would have occurred since they were taken (site infill activity was observed sporadically at many sites up to 2011). The enforcement of permit boundaries would be best achieved through more regular on-site inspections.

Inspections can also be an effective tool in identifying and combating non-compliance of other permit conditions. Available files show at least two occasions where, following inspections, resolutions to issues were documented after issuing warning letters to landowners (Fig 2.12). These issues included accepting waste after the permit expiration date, excessive waste amounts, contamination/fly-tipping and site levelling. However, such letters mostly had no replies or resolutions found on the files. At least three permit renewals are known to have

been refused on the basis of non-compliance of landowners with previous permit terms (due to excessive/contaminated waste and road damage).

Temporal variation of infill coverage was observed over 15 years from aerial photography (OSI, 1995; 2000; 2005; 2010) for each site (n = 133) with GIS software (ArcGIS 10; Fig 2.13). Eleven sites (8 %) had been at least partially infilled (OSI, 2000) before obtaining a WP after 2001. By 2010, 37.5 % of sites which had been refused waste permits (with files available; n = 24) had, nonetheless, been at least partially infilled. It was noted on the permit application file for one site, that there was no option but to grant a permit, due to the site having been completely infilled prior to the permit application being made. As could be expected based on the economic situation at the time (ESRI, 2014), the most dramatic increase in site abundance and total land cover with C&D waste occurred between the aerial photographs published in 2005 and 2010 (OSI, 2005; 2010), the photography for which was generally taken towards the end of the previous year.

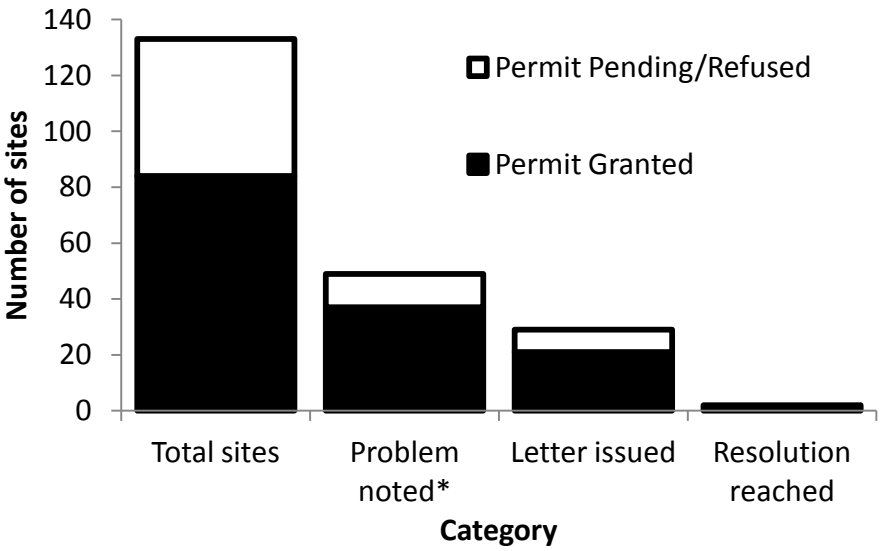

***Fig 2.12**. Problems with non-compliance observed on sites. These data refer only to sites with files (n = 133). *Problems such as fly-tipping or excessive waste were noted through a combination of permit application files (n = 29; as of June 2013) and site visits by principal author (n = 20 additional sites with granted permits).*

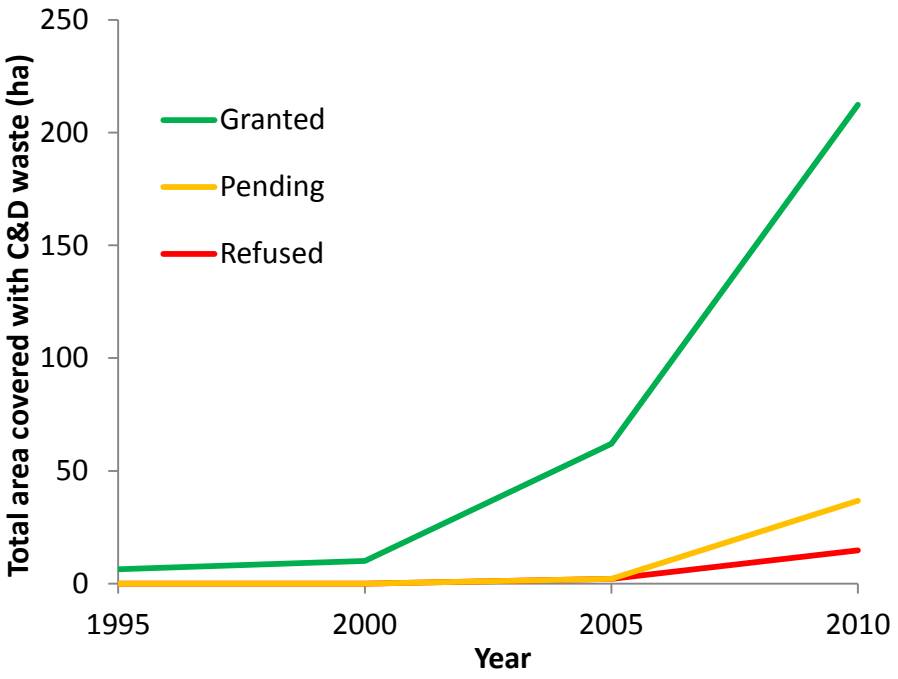

***Fig 2.13**. Stacked line graph showing temporal change in actual area infilled with C&D waste based on aerial photography (OSI, 1995; 2000; 2005; 2010), at Waste Permit, Certificate of Registration and Waste Facility Permit sites in study local authority area, Ireland. This includes only sites for which the application files were available (total n = 133; Granted n = 84; Refused n = 24; Pending decision n = 25). Permit status as of June, 2013.*

Within five kilometres (area with high concentration of documented sites) of the county city limits, it is estimated from aerial photography that there are at least 48 undocumented infill sites for C&D waste, ranging in size from 0.2 ha up to approximately 6ha, with most being < 2 ha (Fig 14). The same area had just 31 legal sites (Fig 2.10). The five largest towns (population) had 14, 7, 19, 28 and 27 such sites found within 5 km. This would indicate that illegal infilling is still a major problem today in Ireland, as has been previously documented in countries such as Hungary, Spain, Italy and the United States (ICF Incorporated, 1995; Symonds Group, 1999; BIO Intelligence Service, 2011). Linear regression analysis found that increasing distance from the urban boundaries was significantly (P = 0.04) negatively correlated with undocumented site density (distance was attributed to 78.9 % of density variation; Fig 2.15). These infill sites are often areas behind farmyards, houses and commercial buildings. Small wet or soft areas are often filled in with C&D waste by landowners without obtaining a

waste permit with the aim of improving the agricultural productivity of the land or creating firm ground. There is little known about the materials used for infilling on these sites, or the depth of the fill. In addition, there was a significant (P = 0.03; Linear regression analysis) positive relationship found between the mean densities of documented and undocumented sites near urban areas (county city and five largest towns). This means that future searches for such sites would likely be most worthwhile near urban centres (i.e. in areas where permitted infill sites would be found).

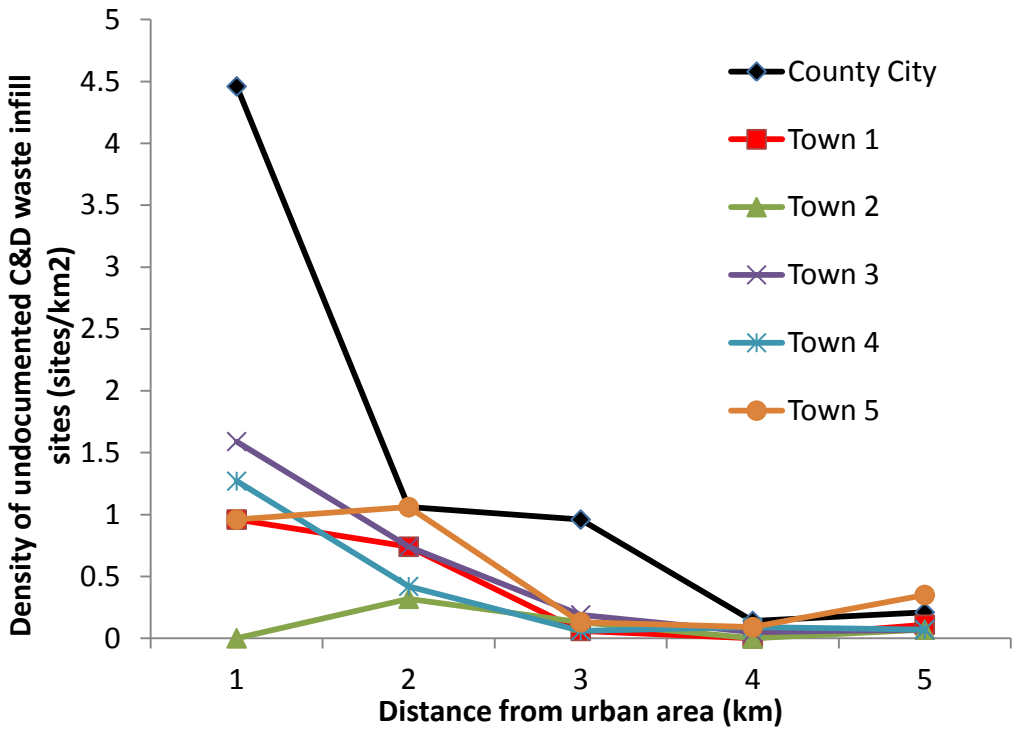

***Fig 2.14**. Density of undocumented sites around the county city and the five biggest towns (population; Central Statistics Office, 2012) in the Local Authority area, Ireland.*

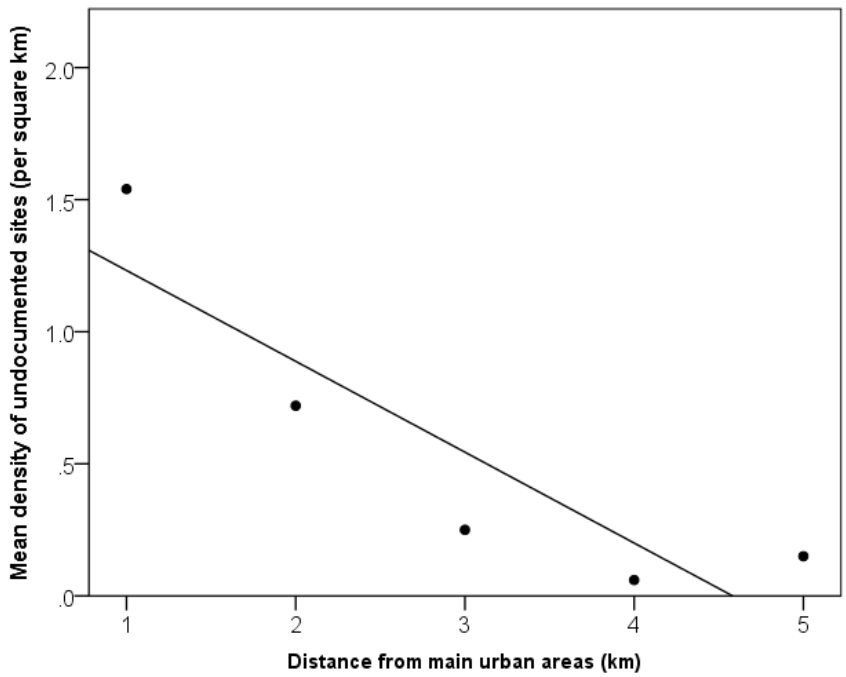

***Fig 2.15**. Scatterplot showing significant (P = 0.04; Pearson correlation, Linear regression analysis) relationship between density of undocumented sites with distance from urban areas (county city and five largest towns) in study county, Ireland.*

The documentation and recording of many aspects relating to C&D waste disposal is known to be lacking internationally (Symonds Group Ltd., 1999, European Commission [DG ENV], 2011), a point that was supported by the available documentation for this study. This may be due to the perceived low environmental priority of the mainly inert waste, but recent research (Staunton et. al., 2014; See Chapters 3 and 4) would suggest that the C&D waste may have a greater environmental impact than once thought, in terms of both ecological and contaminative impacts. Through the course of this case study, some difficulties facing the effective regulation of C&D waste infilling were highlighted. One such difficulty is the under resourcing of Local Authorities to allow frequent site inspections, and to pursue landowners that break the permit terms. This problem exacerbates the situation as landowners may see little or no consequences to ignoring permit terms or not getting a permit at all. Other issues include a lack of awareness among landowners and permit holders regarding the potential harmful effects of using contaminated waste for infilling, or burying prohibited wastes

under the C&D waste, and the problems associated with self-regulation of waste (quantity and constituents) being accepted on site.

The most preferential solution (Council Directive 2008/98/EC) to the problem of C&D waste disposal would be reducing production at source, or at least a reduction in the amount of C&D waste ending up in these unlined disposal or infill sites. The European Council Directive on waste (2008/98/EC) defines waste recovery (under which falls the use of C&D waste as infill material) as any operation "the principal result of which is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfil a particular function". There has been scepticism (Symonds Group Ltd., 1999) as to whether or not infilling would take place at the same rate without such a high availability of C&D waste, and this view was found to be shared by landowners in this study based on collected anecdotal evidence. This would contradict the current theory (Council Directive, 2008/98/EC) that infilling with C&D waste is a recovery operation. It is, rather waste disposal, as the C&D waste would not be replacing virgin infill material. Instead, reducing the availability of C&D waste for disposal (currently thought to be recovery), through recycling, would be a preferred option. Recycling involves the reprocessing of waste into products that can then be used for a purpose (European Council Directive 2008/98/EC), but use for infilling operations is specifically excluded in the Directive. There have been many studies regarding the viability of C&D waste recycling (Lawson and Douglas, 2001; Zhao *et al*., 2010; Coelho and de Brito, 2013; Duran *et al*., 2006; Weil *et al*., 2006; Yuan *et al*., 2011), with the main concerns for its implementation being economic feasibility (Duran *et al*., 2006; Yuan *et al*., 2011) and the quality of the end product aggregates (Bianchini *et al* 2005; Tomas *et al*., 1999; Coelho and de Brito, 2013; Tam *et al*., 2009). It is also thought that infilling lands with C&D waste is, as a low cost activity, a disincentive to the more expensive recycling of the waste (RPS Consulting Engineers, 2006). Duran *et al*. (2005) and Yuan *et al*. (2010) demonstrate that recycling C&D waste would only be viable if the cost of doing so to the producer was lower than the cost of landfilling the waste. Although there is currently a limited market for recycled C&D waste both in Ireland and internationally due to the low cost of raw materials, it has been shown that implementing policy to subsidise the recycling

process, and increased taxes on the use of virgin materials, along with the use of large processing facilities, would be likely to increase recycling rates, and generate a profitable industry (Duran *et al*., 2005; Yuan *et al*., 2010). The recycled (processed) form of the waste can be used as aggregate in road construction, ready mix concrete and other building products (Weil *et al*., 2006; Fong *et al*., 2002; Duran *et al*., 2005; Coelho and de Brito, 2013). Recycling C&D waste has numerous benefits, including a reduced amount of waste used in landfills and reduced demand for raw quarry material, thus decreasing environmental impacts (Duran *et al*., 2005; Weil *et al*., 2006). There is, however, some concern over recycling C&D waste containing harmful chemicals such as lead based paint and polychlorinated biphenyls, and the difficulty and expense of testing and removing contaminated waste may act as a deterrent to recycling (USEPA, 2009).

Currently C&D waste is usually used as a fill material in its unprocessed form, not only in Ireland (Duran *et al*., 2005), but internationally (Lawson and Douglas, 2001; Coelho and de Brito, 2013). The local authority area for this study currently has no dedicated C&D waste recycling plants, and although three mixed waste sorting facilities currently undertake the processing of small amounts of mixed C&D waste (< 3,000 tonnes total per annum; EPA, 2014), the inert fraction of the waste from these facilities is mainly used as infill material on adjacent sites (Waste sorting facility Staff Member, Personal communication). The majority of C&D waste infill locations accept waste directly from construction sites and waste sites via collection vehicles. Waste management plans implemented at a (building) site level are known to aid in the separation and recycling process and increase its efficiency and competitiveness (FÁS Environmental Training Unit, 2002; NCDWC, 2006; Waste sorting facility Staff Member, Personal communication).

There are a number of key recommendations to be made following this study:

- Local authorities in Europe and around the world should adopt the US strategy of ensuring that the proposed site will have a lower environmental impact than other potential sites in the region. Ease of access to the site should be a secondary criterion to this. Any infilling that does occur should have some mitigation plans associated with it, such as, at a minimum, preserving part of the site as intact wetland.

- Ensure that all future infill sites must have a base environmental and ecological survey (flora, fauna, soil, hydrology) carried out prior to infilling. This will ensure a greater understanding of habitat loss, and will allow future site monitoring (permit stipulations could allow for this).
- Local Authorities need enough resourcing to effectively police the sites throughout the infilling procedure, in addition to educating landowners and permit holders on the environmental consequences of ignoring permit stipulations.
- Encourage the recycling of C&D waste by educating personnel on building sites on waste minimisation and separation, and creating a market for the end products through taxation on virgin aggregates and subsidies on the recycling process.
- Carry out wider international studies to identify the distribution of undocumented infill sites, and the frequency with which permit terms are broken on permitted sites.

2.5. Conclusions

This study has, for the first time in a European country, assessed the distribution patterns of C&D waste infill sites at a local scale. It appears that they tend to be located primarily in easily accessed locations, with many of them being very close to urban areas and major roads. Infill sites were also found to be common on aquifers of extreme vulnerability and immediately adjacent to designated protected habitats (SACs). Most of the applications for the waste permits contain little or no information on local habitats, any likely ecological effect of the waste and suitable mitigation plans for the sites. Undocumented infill sites appear to follow similar dispersal patterns as permitted sites near urban areas, but with a higher density, suggesting that the problem of illegal infilling is a common occurrence. Infill sites were found to regularly break the permit conditions of waste volume and area, with some being infilled prior to receiving a permit. In addition, C&D waste infill was found to be frequently contaminated with

unpermitted substances. The limited resources of the local authorities mean they are struggling to effectively monitor and police infill sites and prosecute offending landowners. However, site inspections by local authority personnel were found to be effective in some situations, resulting in a number of resolved problems. The findings of this study can be used by all local authorities and countries to encourage and help in the development of a long term strategic approach to C&D waste disposal.

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Chapter 3

Challenges in assessing ecological impacts of construction and demolition waste on wetlands: A case study

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3. Challenges in assessing ecological impacts of construction and demolition waste on wetlands: A case study

3.1. Abstract.

Although wetlands are of ecological and economic importance, they continue to be lost to anthropogenic activities such as infilling. The impacts of wetland infilling with construction and demolition (C&D) waste on wetland plant and dipteran (Insecta: Diptera) communities were examined. Areas of wetland infilled with C&D waste compared to non-infilled areas had: a) higher soil pH and lower soil moisture / organic content; b) a relatively higher percentage of ruderal plant communities; c) relatively fewer dipteran families that were wetland specialist, gall-forming, parasitic and haematophagous; d) relatively lower abundances and species richness of marsh flies (Diptera: Sciomyzidae). Challenges encountered during this study included locating C&D waste sites; obtaining permission from landowners to undertake this study; frequent damage and theft of equipment due to human interference, machinery and infilling activity. Given the current paucity of data regarding the ecological impacts of infilling with C&D waste on wetlands and the considerable challenges with undertaking such studies, we make recommendations for appropriate site selection and monitoring at C&D waste infill sites.

3.2. Introduction

3.2.1. Wetlands

Wetlands are considered as some of the most ecologically and economically important habitats worldwide. Covering between seven and ten million km^2 globally, they provide many important ecosystem services (Costanza *et al*., 1997; Keddy, 2000; Lehner and Döll, 2004), including the provision of essential habitats for wetland plant and invertebrate communities, water filtration and flood control. However, wetlands have been and continue to be lost at significant rates: two-thirds of European wetlands were lost during the 20th century due to anthropogenic activities (EC, 1995) such as draining, dredging and infilling (Mitsch and Gosselink, 2007) with agriculture being one of the main driving forces behind the loss (Chen *et al*. 2012). This is not surprising given that wetlands are frequently perceived as land with no direct economic benefit to the landowner. Infilling with construction and demolition waste (Poon, 2001; Shen *et al*., 2004) is, therefore, seen as a means of creating improved agricultural grassland by covering the infill with topsoil or developing dry, elevated sites for building purposes.

3.2.2. Construction and demolition waste

Construction and demolition (C&D) waste can be described as waste that is produced as a result of the construction, demolition or renovation of structures (Shen *et al*., 2004; USEPA, 2009). It is composed of a mix of wastes from building sites, including concrete, wood and asphalt (EPA, 2009; Fischer and Werge, 2009; Poon *et al*., 2001; Williams, 1998). Although approximately 870 million tonnes (32.9% of total waste) of C&D waste were produced in EU countries in 2008 (Eurostat, 2011), detailed information regarding the disposal of the waste is not currently available (European Commission DG ENV, 2011) given that EU countries frequently categorise infilling as C&D waste recycling. Nevertheless, some EU countries (Spain, Hungary and Ireland) have documented problems with illegal disposal of C&D waste (European Commission DG ENV, 2011; EPA staff member, Pers. Comm.) in unregulated fill sites.

Information on the regulation of C&D waste infill in Ireland is presented in Table 3.1. Prior to 2001, municipal landfills were the only legal sites at which C&D waste could be placed. However, with the waste being viewed as mainly inert, it was often used as unregulated fill material (EPA, 1996; EPA staff member, Pers. Comm.). Post-2001 Waste Permits (WP) were obtained for many of these unregulated sites, usually without any ecological assessment, so that infilling could continue. The more recently introduced Certificates of Registration (COR) which require submission of biodiversity details of the site, give no indication of the level of ecological detail required for the granting of the permit. Given this situation, the loss, due to C&D infilling, of unprotected Irish wetlands and their associated biota is likely to be still taking place.

Table 3.1. *Regulations relating to Construction and Demolition (C&D) Waste infilling in the Republic of Ireland. ta^{-1} = tonnes per annum. na = not applicable. EIA = Environmental Impact Assessment. AA = Appropriate Assessment.*

	C&D waste disposal permits [a, b, c, d, e, f]	**Outcomes** [a, b, c]	**Ecological survey details** [b, c, d, e, f]
Pre 2001	No C&D specific disposal sites. C&D waste was directed to municipal landfills	C&D waste often used as unregulated fill material. Many of these sites granted WPs post 2001 to continue infilling	Na
2001 - 2008	<5,000 ta^{-1}: Waste permits (WP) granted by local authority	Most inert C&D waste was disposed on WP sites	Ecological information not essential - Environmental survey (no details given) decided on a case-by-case basis in situations where pollution is likely, or site is near/in Natura 2000 site
	>5,000 ta^{-1}: Waste License granted by EPA	Few Waste Licenses granted with result that most inert C&D waste was disposed on WP sites	Environmental survey (EIA) required for waste license. (See EPA Advice Notes on Current Practice[g])
2008 - present	<10,000 ta^{-1}: Certificate of Registration (COR) granted by local authority	Most inert C&D waste disposed on COR or WFP sites	Application requires details of biodiversity on site. No information on minimum level of ecological detail required. EIA required if pollution is likely - this is decided on a case-by-case basis or if waste >25,000 tonnes. AA[h] required if there are potential impacts on NATURA 2000 site
	10,000 - 50,000 ta^{-1}: Waste Facility Permit (WFP) granted by local authority		
	>50,000 ta^{-1}: Waste License granted by EPA	Most inert C&D waste disposed on COR or WFP sites	EIA required. AA[h] required if there are potential impacts on NATURA 2000 site

[a] EPA, 1996. [b] EPA staff member, Pers. Comm. [c] Local authority staff member, Pers. Comm. [d] Statutory Instrument No. 165 of 1998. [e] Statutory Instrument No. 821 of 2007. [f] Office of Environmental Enforcement, 2008. [g] CAAS Environmental Services Ltd., 2003 [h] European Commission, 2002.

With the exception of a single publication by Gabrey (1997) which found that C&D waste had no significant impacts on bird populations in the USA (in the context of birds as hazards to nearby airports), the ecological impacts of infilling wetlands with C&D waste have been poorly studied. Wetland sites infilled with C&D waste are, at best, challenging sites to complete ecological investigations, for a number of reasons. Landowners may refuse requests to undertake site surveys (noted by Krause *et al*., 2013 when undertaking stream investigations) due to the possibility, in this case, of an ecological surveyor discovering hazardous, non-C&D waste material. On the other hand, C&D waste sites frequently have open access and are subject to constant disturbance, not only from machinery dumping and spreading the C&D waste but from illegal fly-tipping activities. While the authors quickly became aware of these challenges early in this study, we nevertheless persisted with our investigations in the belief that quantitative data, in the form of a case study, would go at least some way in highlighting the ecological effects of infilling wetlands with C&D waste, given the paucity of knowledge in this field. These data can bring to light potential ecological impacts on wetlands of C&D waste with a view to informing policy changes for future site selection and monitoring. With this in mind, we concentrated on wetland biological groups such as plants which are sensitive to chemical changes in their environment (LaPaix *et al*.2009; Pardo *et al*., 2011) and Diptera (families and morphospecies), shown to be influenced by vegetation structure (Hughes *et al*., 2000; King & Brazner, 1999; Whiles and Goldowitz, 2001). In particular, we identified marsh flies (Diptera: Sciomyzidae), to species level since they have been shown to reflect a range of wetland conditions (Murphy *et al*., 2012; Speight, 1986; Williams *et al*., 2009, 2010). While plants are frequently used in isolation to assess habitats, we included invertebrate groups in this study given that, apart from iconic invertebrate species such as butterflies, policy makers can often be unaware of problems associated with general invertebrate conservation (Cardoso *et al*., 2011).

This study presents a description of nine wetland sites which have been affected by the infilling of C&D waste. The objectives of the study are to compare, for the first time, plant and dipteran communities on the C&D infilled and non-infilled portions of wetlands. Our hypothesis is that plant and dipteran community

composition will be significantly different on C&D infilled and non-infilled portions of wetlands. In addition, we identify problems currently associated with ecological site investigations at C&D infill sites with a view to developing recommendations for appropriate site selection and monitoring.

3.3. Methods

3.3.1. Study area

Nine sites (Table 3.2), located in County Galway (Fig. 3.1) in the west of Ireland were investigated for this study. Eleven sites were originally selected for the present study. However, two of these sites had to be abandoned within weeks of starting due to repeated vandalism and theft of invertebrate sampling equipment. Sites were chosen from all County Galway sites for which permits were held for the disposal of C&D waste. They were selected on the basis of proximity to each other so that aerial invertebrate samples could be collected from all sites on the same day, thereby reducing the influence of weather conditions on invertebrate catches. Most sites were chosen in areas to the north of Galway city where there is a concentration of wetlands. Sites were selected from those wetlands which were partly infilled with C&D waste to facilitate comparisons between the infilled and non-infilled portions of the wetlands. Habitat classification was carried out on the selected sites following Fossitt (2000).

The nine sites (Fig. 1; Table 2) consisted of two (WG1 and WG2) wet grassland sites (soil pH>7), two (SW1 and SW2) reed & large sedge swamp sites (soil pH>7) and five (CB1–CB5) cutover raised bog sites (soil pH<7). Total wetland sizes ranged from 9ha to 169ha (estimated from aerial photography). One site (WG1) was situated within an EU designated Special Area of Conservation (SAC) on the River Clare. All sites had already been partly infilled with C&D waste when this study began, with varying levels of infilling activity being carried out during the study period.

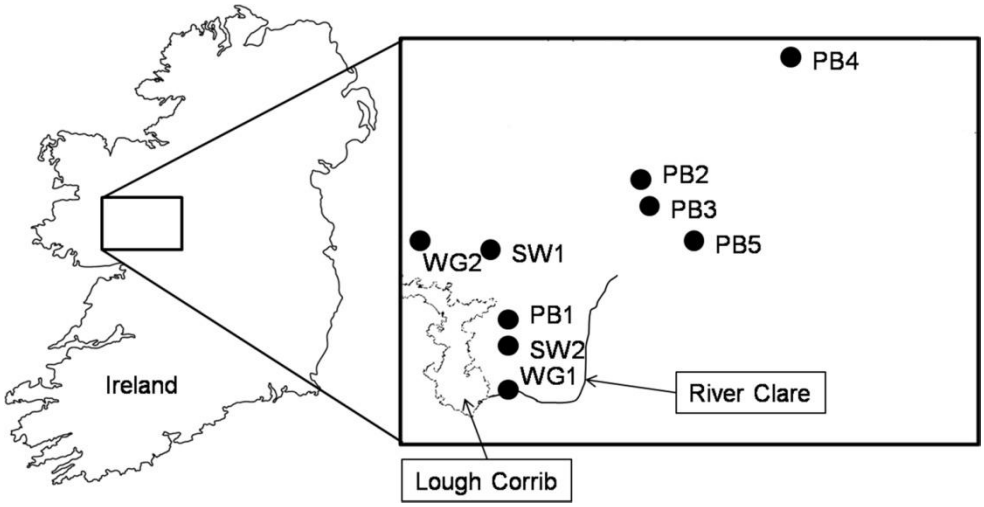

***Fig 3.1**. Location of wetland study sites within County Galway, Ireland*

3.3.2. Sampling methods

Diptera were sampled in 2009 and 2010 while vegetation surveys and soil sampling were undertaken in 2010. However, due to infilling activity and discontinuation of access permission to sites CB5 and SW2 respectively at the end of 2009, vegetation surveys and soil sampling took place on seven sites only. Vegetation surveys were carried out on sites WG1, WG2, SW1 and CB1-CB4 in August 2010. Sampling in the wetland using three 0.5m x 0.5m quadrats (Bullock, 2006), 5m apart, was restricted to 5m from the edge of the infill and, in the infill, to 5m from the edge of the wetland. This sampling strategy was limited by the size of the smallest site with other sites being sampled in the same manner for comparative purposes. In addition, depth of water became greater in some of the wetlands with distance from the infill and safety considerations prevented sampling in these areas. Nevertheless, given the abrupt changes in plant communities that can be seen at the interface between the infill and wetland (Fig. 3.2), the vegetation data recorded gives a good indication of differences in plant communities at the infill and wetland interface.

All plant species within each quadrat were identified using Rose and O'Reilly (2006) and Webb *et al*. (1996). Percentage cover of each plant species, bryophytes (bryophytes were not identified to species level, but were dealt with as a group due to time constraints), dead vegetation and bare ground were recorded. Within

each quadrat, four measurements were taken randomly with a ruler for both vegetation height (maximum height from ground of resting vegetation) and vegetation length (length of longest plant when stretched out), as measurements of structural complexity for use in data analysis (Williams, 2010). Ellenberg indices (Hill *et al*., 1999), as corrected for use in the British Isles by Hill *et al*. (2000), were used as additional surrogate environmental variables, and were calculated following Williams *et al*. (2011). Ellenberg indices are based on the plant community data and can be used to indicate soil parameters (moisture, pH and nitrogen content) and light intensity (reflecting sward structure and density). Ellenberg values (moisture and pH) were also compared with measured field soil parameters. Ellenberg values provide an easy method of estimating such parameters without being influenced by temporal weather variation.

Fig 3.2. *Example of interface between infill and wetland*

Using a Dutch auger (Eijkelkamp), soil samples at each quadrat (ca. two kilograms) were taken in 2012 to a depth of 20cm. C&D waste which was frequently compacted by heavy machinery was difficult to penetrate preventing the extraction of samples at lower depths. Moisture content (expressed as a percentage of the wet weight), mass loss-on-ignition (expressed as a percentage of the dry weight) and pH (using soil suspensions) were determined according to

British Standards (BSI, 1990). Results from individual samples were averaged (mean if normal distribution obtained, otherwise median) for the infill and for the wetland zones of each site for comparison.

Aerial invertebrates (Diptera) were collected using pan traps (Southwood, 1978) at all nine sites in 2009 and from the remaining seven sites in 2010. All sciomyzids were removed and identified from these samples. In addition, aerial invertebrates collected in 2009 were identified to morphospecies level for those seven sites in which plant surveys were undertaken in 2010 to allow comparison of the plant and aerial invertebrate data. Each pan trap consisted of a white plastic container (20cm diameter x 10cm high) placed within a similar container fixed to a wooden post set at 50cm (allowing for flood events) above ground level (Southwood 1978; Campbell and Hanula, 2007). One pan trap was placed in the centre of each vegetation quadrat. While it could be argued that some dipteran species could move between infill and wetland trap areas, any differences in data for dipteran community composition are likely to be real differences reflecting the nature of the habitats. A 25% solution of ethylene glycol (preservative) was added to the pan trap (filled to two centimetres from rim) in addition to a small amount of Ecover® washing up liquid, which was used as a surfactant. The traps were emptied weekly (July 14 to October 13 in 2009; May 6 to September 30 in 2010) and trapped invertebrates were collected by straining the trap contents through a fine nylon mesh (0.5mm). All samples were then preserved in a 70% ethanol solution. Sciomyzidae were identified to species level (Rozkošný, 1984, 1987) for all dates, and all dipteran individuals were identified to family (Oosterbroek, 2006; Unwin, 1981) for three sampling dates in 2009 (14 July; 1 September; 13 October). Taxonomic minimalism reduces time spent on species identification, allowing more samples to be analysed (Beattie and Oliver 1994), while still being a useful method to assess biodiversity (Rivers-Moore and Samways, 1996). Groups with different morphological features were identified within dipteran families and treated as separate morphospecies (Beattie and Oliver, 1994). Adult sciomyzids were identified to species level since they are known to remain close to where they eclose and therefore, reflect different types of wetland conditions (Williams *et al*., 2010).

Aerial invertebrates were also sampled at one site (WG2) using a sweep net (50cm diameter x 67 cm bag depth and 30.5cm handle length) (Williams *et al*., 2009) every two weeks in 2010 (19th May to 22nd September) allowing comparison of catches caught by sweep-netting with the pan trap method. Eight parallel sweep paths (ten metres long with a two metre buffer zone between each) were marked out using bamboo canes on both the wetland and the infill. Vegetation to the east of each path was swept in the standard Fig of eight motion (ca. 1m wide), and this was carried out by the same person using a consistent walking pace and sweeping speed. The invertebrates in the sweep net were euthanized for each sweep path in the field (with each sweep path being a separate sample) by placing in a kilner jar (12cm diameter x 30cm) with ethyl acetate (99.5%). Samples were preserved in 70% ethanol and sciomyzids were identified to species level (Rozkošný, 1984, 1987). Environmental variables measured at the time of sampling were vegetation height, length of outstretched vegetation (both measured beside each sweep path; using both provides information on sward structure), wind speed, humidity (Skywatch® Atmos by JD Industries), light intensity (Hanna Lux meter HI97500) and nebulosity (visual percentage estimate). Uneven surface topography prevented sweep netting at the other sites.

3.3.3. Statistical analysis

Various statistical procedures were carried out on the collected data to assess if there was a significant difference between the biota of the of wetland area infilled with C&D waste compared to non-infilled wetland area. Multi-Response Permutation-Procedure (MRPP) was used for observing the strength of grouping variables (habitat type and site) for multivariate datasets (Meilke and Berry, 2001). Non-metric multi-dimensional scaling (NMS) ordinations which do not assume multivariate normality were used to compare plant and dipteran communities of the wetlands and C&D waste infill (Kenkel and Orloci, 1986). Indicator Species Analysis, a method for observing the association of a species with a particular grouping variable, in this case, habitat (Dufrêne and Legendre, 1997), was also undertaken. Shannon's entropy is used (instead of Shannon's diversity index) as entropy has been shown to be more useful, giving a value for the uncertainty in the data, rather than true diversity (Jost, 2006). Minitab®

Statistical Software (version 16) was used for univariate statistical analysis, and PC-ORD (version 6) was used for multivariate analyses (McCune and Grace, 2002; McCune and Mefford, 1999).

3.4. Results

3.4.1. Soils and plant communities

Overall there were significant differences found between the soil parameters and plant communities of C&D infill and wetland. When all wetland sites were combined for analysis, mean soil pH was significantly (t=5.71, $P<0.05$) greater on the infill (7.94) than the wetlands (6.41). In addition, median percentage soil moisture content (Table 3.2) was significantly (T<0.1, $P<0.05$) lower on the infill (15.35%) than in the wetland (80.69%) as was (t=11.34, $P<0.05$) mean percentage soil mass loss-on-ignition (5.03% on infill and 70.95% on wetland). Although the cutover raised bog wetlands were all acidic (pH<7) with the remaining wetland habitats being alkaline (pH>7), nevertheless, for each site studied, the pH of the C&D waste substrate was significantly ($P<0.05$) higher than the median pH of the original wetland (Table 3.2).

A total of 94 plant species were recorded at the seven sites in 2010 (n=42 quadrats), with median plant species richness and Shannon's entropy being significantly ($P<0.05$) higher on the infill than on wetlands regardless of wetland type (Table 3.3). However, there was no significant difference ($P>0.05$) in plant species evenness between infill and wetland. Following separation of sites (according to wetland soil analysis) into acidic (pH<7) or alkaline (pH≥7) wetlands, the plant data show that variance in plant species composition that was attributable to habitat status (i.e. wetland versus C&D waste) was slightly higher on acidic sites (15% of variance) than on alkaline sites (14% of variance), based on MRPP (Table 3.4). Table 2 shows the most dominant plant species for each site (wetland and C&D waste infill). Plant indicator species analysis was performed on the acidic and alkaline sites separately using the Monte-Carlo test of significance (Table 3.5). Eleven plant species were found to be significant indicators of C&D waste on acidic sites, and six on alkaline sites, with *Agrostis*

stolonifera L. having the highest percentage of perfect indication on both. Of the six indicator plant species of C&D waste on alkaline sites, four (*A. stolonifera, Cerastium fontanum* Baugm, *Lolium perenne* L. and *Ranunculus repens* L.) were also listed as indicators of C&D waste on acid sites (*Table 5*). Four plant species (*Calluna vulgaris* (L.) Hull, *Erica tetralix* L., *Molinia caerulea* (L.) Moench and *Potentilla erecta* (L.) Rauschel) were significant indicators of wetland on acidic sites, with *M. caerulea* having the highest percentage of perfect indication (Table 3.5). Alkaline sites, which consisted of more variable wetland types, were without significant wetland indicator species.

***Table 3.2**. Brief description of the nine wetland and infill study sites in Co. Galway, Ireland. Site code explanation: WG = Wet Grassland, SW = Swamp, CB = Cutover Raised Bog, nd = no data.*

Site code	Total site area with permission for infilling (ha)	Percent of total permitted area with infill	Mean soil pH Wetland [T, a] (Infill)	Mean soil mass loss on ignition % Wetland [T, a] (Infill)	Median soil moisture content % Wetland [W, a] (Infill)	Two most dominant wetland plant species [b]	Two most dominant infill plant species [b]
WG1	2.87	89.89	7.65* ± 0.11 (8.10 ± 0.06)	32.61* ± 5.74 (2.01 ± 1.26)	64.23* (14.04)	*Agropyron repens, Deschampsia cespitosa*	*Agrostis stolonifera, Festuca rubra*
WG2	0.88	32.95	7.74* ± 0.06 (8.00 ± 0.04)	37.39* ± 13.89 (2.50 ± 0.03)	77.86** (14.67)	*Carex disticha, Cardamine pratensis*	*Agrostis stolonifera, Trifolium repens*
SW1	6.62	15.56	7.13** ± 0.14 (8.01 ± 0.15)	63.45* ± 14.83 (2.31 ± 0.63)	69.28* (14.02)	*Cladium mariscus, Iris pseudacorus*	*Agrostis stolonifera, Cirsium arvense*
SW2	12.87	4.58	nd	Nd	nd	*Cladium mariscus Phragmites australis*	*Agropyron repens, Sonchus asper*
CB1	5.72	47.55	6.56* ± 0.63 (7.83 ± 0.15)	85.53** ± 1.77 (1.94 ± 0.60)	89.71** (16.64)	*Molinia caerulea, Calluna vulgaris*	*Agrostis stolonifera, Trifolium pratense*
CB2	9.05	4.09	5.01** ± 0.25 (7.89 ± 0.01)	95.93** ± 1.73 (6.23 ± 1.02)	84.27** (27.74)	*Calluna vulgaris, Carex rostrata*	*Agrostis stolonifera, Centaurea nigra*
CB3	4.03	70.22	6.08* ± 0.49 (7.54 ± 0.27)	94.79** ± 2.19 (12.87 ± 1.15)	89.95** (32.43)	*Calluna vulgaris, Eriophorum angustifolium*	*Agrostis stolonifera, Circium arvense*
CB4	1.53	17.00	4.53** ± 0.19 (8.21 ± 0.26)	86.91** ± 1.05 (7.28 ± 3.28)	80.68** (15.35)	*Molinia caerulea, Succisa pratensis*	*Agrostis stolonifera, Plantago lanceolata*
CB5	2.07	71.01	nd	Nd	nd	*Molinia caerulea, Calluna vulgaris*	*Trifolium repens, Holcus lanatus*

* Significant difference between infill and wetland data with (P<0.05) and ** (P<0.005). [W] Wilcoxon signed rank test used. [T] paired t-test used. [a] Mean ± standard deviation calculated from three samples of soil from both infill and wetland areas on each site. [b] Most dominant species based on percentage cover.

***Table 3.3**. Species richness, species evenness and Shannon's entropy of all sites (only WG2 for sweep net data) for vegetation, Sciomyzidae and dipteran families and morphospecies. na = not applicable (due to zero collections).*

Dataset	**Median species richness wetland [W, a] (infill)**	**Median species evenness wetland [W, a] (infill)**	**Median species entropy wetland [W, a] (infill)**
2010 Vegetation	10.00 ± 4.57** (17.00 ± 5.54)	0.74 ± 0.14 (0.75 ± 0.08)	1.66 ± 0.56** (2.24 ± 0.14)
2009 (pan trap) Sciomyzidae	2.00 ± 1.28 (1.00 ± 1.64)	0.45 ± 0.43 (0.00 ± 0.45)	0.56 ± 0.40 (0.00 ± 0.53)
2010 (pan trap) Sciomyzidae	1.00 ± 0.39 (na)	0.00 ± 0.37 (na)	0.00 ± 0.26 (na)
2010 (sweep net) Sciomyzidae	4.00 ± 1.30* (0.00 ± 0.92)	0.79 ± 0.09* (0.00 ± 0.46)	0.63 ± 0.05* 0.00 ± 0.28
2009 Dipteran family	24.00 ± 3.45 (25.00 ± 4.38)	0.81 ± 0.07 (0.77 ± 0.08)	2.61 ± 0.28 (2.44 ± 0.28)
2009 Dipteran morphospecies	48.00 ± 12.08 (44.00 ± 12.07)	0.81 ± 0.07 (0.77 ± 0.09)	3.26 ± 0.40 (2.91 ± 0.42)

* Significant difference between infill and wetland data with (P<0.05) and ** (P<0.005).
[W] Wilcoxon signed rank test used. [a] Median ± standard deviation calculated from all sites.

Non-metric multi-dimensional scaling (NMS) ordinations after McCune and Grace (2002) were performed with the plant data (Fig. 3.3a), with soil moisture, loss-on-ignition and pH included as vectors, resulting in a 3-dimensional solution. The plant community ordination (Fig. 3.4) showed the C&D waste infill communities to be tightly clustered indicating a high similarity between these sites regardless of the type of wetland which had been infilled with C&D waste. The wetland points, however, were more dispersed suggesting a higher variation between wetland plant communities. For the plant community data, axis 1 was most strongly correlated (r^2=.446) with Ellenberg moisture indicating the importance of moisture in determining plant community composition (Table 3.7).

Table 3.4. *Multi-Response Permutation-Procedure (MRPP) for vegetation, Sciomyzidae and dipteran families and morphospecies (Distance measure: Sørensen). Chance corrected within group agreement is a measure of within group homogeneity and P-values were assessed by permutation. Alkaline = sites with wetland soil pH≥7; Acidic = sites with wetland soil pH<7. na = not applicable (analysis could not be performed as all sciomyzids for 2010 were captured on wetland area).*

Dataset	**Grouping variable**	**Chance-corrected within-group agreement (*A*)**	***P***
2010 Vegetation	Infill v Wetland (alkaline)	0.14	2×10^{-5}
	Infill v Wetland (acidic)	0.15	3×10^{-6}
	Between site (alkaline)	0.19	7×10^{-5}
	Between site (acidic)	0.07	0.01
2009 Dipteran Family	Infill v Wetland (alkaline)	0.04	[a]
	Infill v Wetland (acidic)	0.06	3×10^{-4}
	Between site (alkaline)	0.32	5×10^{-8}
	Between site (acidic)	0.14	3×10^{-6}
2009 Dipteran Morphospecies	Infill v Wetland	0.04	2×10^{-5}
	Between site	0.21	10^{-8}
2009 sciomyzid pan traps	Infill v Wetland	0.07	5×10^{-3}
	Between site	0.11	0.03
2010 sciomyzid pan traps	Infill v Wetland	na	Na
	Between site	0.02	[a]
2010 sciomyzid sweep net	Infill v wetland	0.14	5×10^{-3}

[a] not significant at *P*>

Table 3.5. *Indicator species analysis results for vegetation data from 2010 (n=42 quadrats). (Monte-Carlo randomised test, 4999 permutations). IV = percent perfect indication.*

Max. Group		**Indicator species**	**IV**	***P* (4,999 permutations)**
Acidic sites only	Infill	*Agrostis stolonifera*	99.7	2×10^{-4}
		Cerastium fontanum	69.6	0.031
		Circium arvense	70.0	0.002
		Holcus lanatus	86.1	2×10^{-4}
		Lathyrus pratensis	41.7	0.037
		Lolium perenne	66.7	0.008
		Plantago lanceolata	54.8	0.019
		Polygonum amphibium	41.7	0.037
		Ranunculus repens	73.8	6×10^{-4}
		Taraxacum officinalis	41.7	0.036
		Trifolium repens	66.7	0.002
	Wetland	*Calluna vulgaris*	58.3	0.006
		Erica tetralix	38.1	0.002
		Molinia caerulea	74.5	4×10^{-4}
		Potentilla erecta	71.2	6×10^{-4}
Alkaline sites only	Infill	*Agrostis stolonifera*	99.8	4×10^{-4}
		Bryophytes[a]	83.8	0.001
		Cerastium fontanum	55.6	0.031
		Festuca rubra	99.4	4×10^{-4}
		Lolium perenne	66.7	0.008
		Ranunculus repens	88.9	6×10^{-4}
	Wetland	No indicators with $P<0.05$	-	-

[a]More indicator species would likely have been found if Bryophytes were identified to species level.

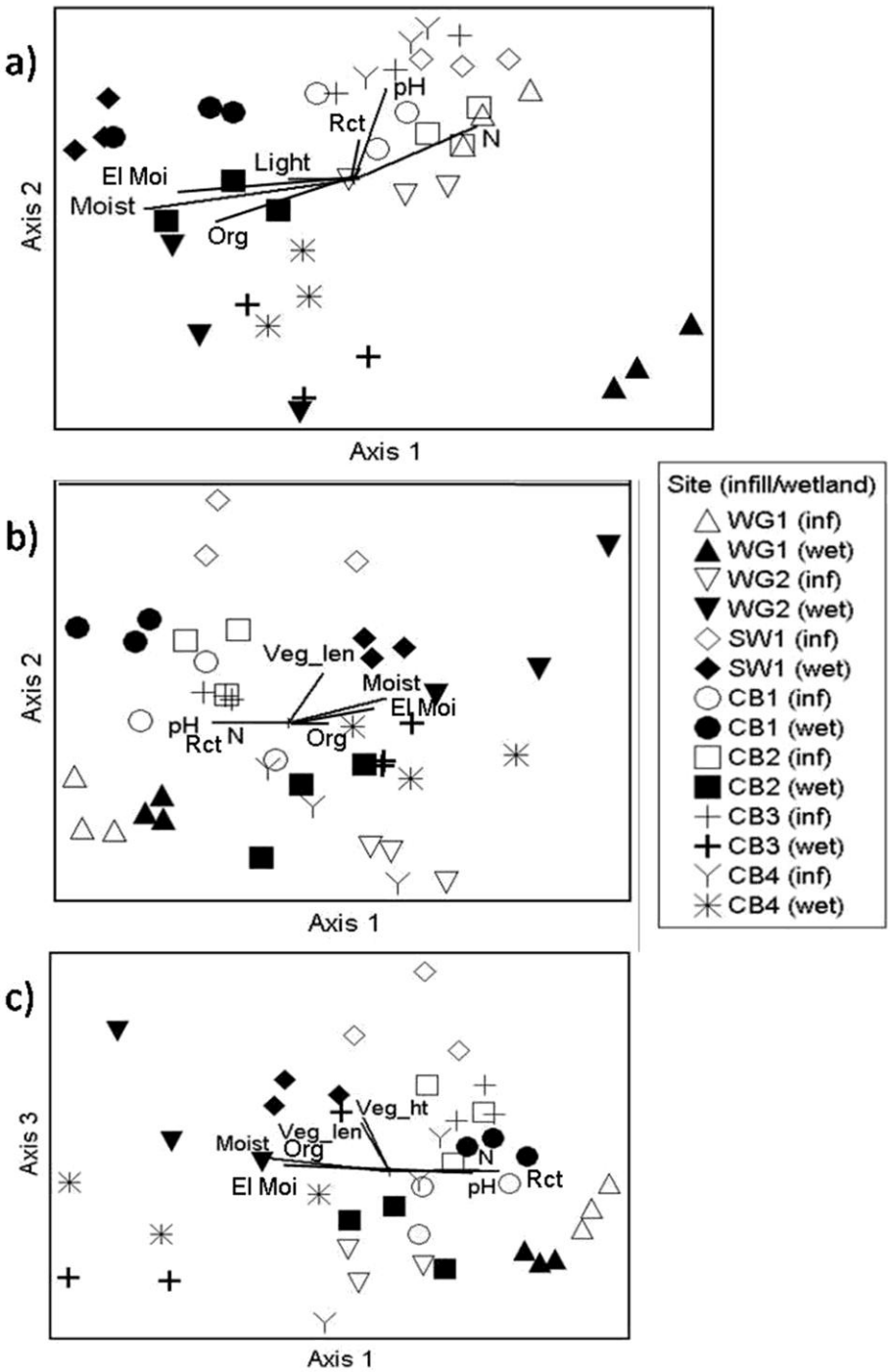

Fig 3.3. *Non-metric multi-dimensional scaling (NMS) ordinations of sites in plant species space (a), dipteran family space (b) and dipteran morphospecies space (c). a) shows the 2010 plant community data (48 iterations, stress of 12.316). b) shows the 2009 dipteran family data from pan traps (45 iterations, final stress of 11.584). c) shows the 2009 dipteran morphospecies data from pan traps (47 iterations, final stress of 11.711). Distance measure: Sørensen, random starting configuration, three-dimensional solutions with orthogonality of 100%, final instability of <0.001. Coefficients of determination for the correlations between*

ordination distances and distances in the original n-dimensional space were: a): axis 1 = 0.287, axis 2 = 0.288, axis 3 = 0.154, b) axis 1 = 0.516, axis 2 = 0.224 and axis 3 = 0.055 and c) axis 1 = 0.565, axis 2 = 0.117 and axis 3 = 0.131. Environmental variables are overlaid as vectors. Light = Ellenberg light, N = Ellenberg soil nitrogen, El Moi = Ellenberg soil moisture, Rct = Ellenberg soil reaction (pH), Veg_len = vegetation length, Veg_ht = vegetation height, Moist = percentage soil moisture, pH = soil pH, Org = percentage soil mass loss-on-ignition (organic content), wet = wetland and inf = infill.

3.4.2. Dipteran communities

Forty-four dipteran families were identified from a total of 10,688 individuals collected using pan traps on three sampling dates in 2009 (14 July; 1 September; 13 October) across seven sites (CB1 – CB4, SW1, WG1 & WG2). There was no significant difference in median morphospecies richness, evenness or Shannon's entropy for any dipteran families identified (Table 3.3). Following separation by wetland soil pH (as with plant data above), site-specific differences accounted for 32% of variation in dipteran family composition for alkaline sites, and 14% for acidic sites whereas habitat status (infill v wetland) accounted for only 4% and 6% of the variance, respectively based on MRPP (Table 3.4). Table 3.6 shows the significant indicator dipteran families for C&D waste (six families) and wetlands (four families). The families Chloropidae and Phoridae had the highest percentage of perfect indication for infill at 73.7% and 66.8%, respectively. For wetlands, the two best indicators were Culicidae and Chironomidae, with values of 71.8% and 69.5%, respectively. The Family Sciomyzidae was found to have a percentage of perfect indication of 26.1% (P = 0.083) as indicators of wetlands. This low percentage (and higher P value) is likely due to low abundances, especially on the cutover raised bog sites. Similar to the plant indicator species, there was no overlap in indicator morphospecies or sciomyzids between infill and wetlands.

Two hundred and seven dipteran morphospecies were identified. Site specific factors were the most important factor in accounting for differences among morphospecies (ca. 21%) based on MRPP analysis of all seven sites (Table 3.4). The habitat status (i.e. infill versus wetland) accounted for ca. 4% of

morphospecies compositional differences among the data-sets. There were seven significant indicator morphospecies for C&D waste (within Calliphoridae, Carnidae, Chloropidae, Muscidae, Phoridae and Sepsidae) and nine for wetlands (within Cecidomyiidae, Chironomidae, Lauxaniidae, Psychodidae, Sciaridae and Tachinidae). There was a lack of obvious clustering in the dipteran family and morphospecies NMS ordinations (Fig. 3.3b and Fig. 3.3c). This suggested little advantage to using morphospecies, when compared to family data alone. For dipteran family (Fig. 3.3b) and morphospecies (Fig. 3.3c) data-sets, soil moisture (r^2=0.298) and Ellenberg moisture (r^2=0.466) respectively were the most strongly correlated parameters with axis 1 (Table 3.7).

Table 3.6. *Indicator species analysis results pan trap dipteran family data from 2009 (n=42 pans), pan trap sciomyzid data from 2009 (n=54 pans) and 2010 (n=42 pans), and sweep net sciomyzid data (n=16 paths) for 2010 (Monte-Carlo randomised test, 4999 permutations). IV = percent perfect indication.*

Dataset	**Indicator species / Family**	**Max. group:**	**Abundance in Max. grp.**	**IV**	***P* (4999 permutations)**
2009 Pan trap dipteran families	Anisopodidae	Infill	17	38.5	0.040
	Carnidae		931	64.4	0.017
	Chloropidae		609	73.7	0.001
	Muscidae		286	62.6	0.032
	Phoridae		510	66.8	2×10^{-4}
	Sepsidae		78	62.4	0.010
	Cecidomiidae	Wetland	113	67.7	0.003
	Chironomidae		421	69.5	0.036
	Culicidae		144	71.8	0.001
	Tachinidae		24	47.9	0.015
	Sciomyzidae[a]		23	26.1	0.083
2009 Pan traps sciomyzid	No indicators with P<0.05	Infill	-	-	-
	Tetanocera robusta	Wetland	82	61.4	0.048
2010 Pan traps sciomyzid	No indicators with *P*<0.05	Infill	-	-	-
	Tetanocera robusta	Wetland	12	52.4	0.001
2010 Sweep net sciomyzid	No indicators with *P*<0.05	Infill	-	-	-
	Pherbina coryleti	Wetland	41	97.6	2×10^{-4}
	Ilione albiseta		39	83.2	8×10^{-4}
	Tetanocera ferruginea		11	80.2	0.006

[a] Although Sciomyzidae were not a significant indicator family, some sciomyzid species were.

Table 3.7. *Pearsons'correlation coefficients between environmental variables and axes of NMS ordinations a) 2010 plant community data-set, b) 2009 dipteran family data-set, c) 2009 dipteran morphospecies data-set, d) 2010 sciomyzid data-set.*

Data-set	**a) 2010 plants**			**b) 2009 dipteran family**			**c) 2009 dipteran morphospecies**			**d) 2010 sciomyzid**
Axis	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**
Vegetation height	.094	.033	.213	.123	.201	.077	.094	.032	.216	.045
Vegetation length	.104	.017	.198	.118	.254	.138	.103	.016	.200	.205
Ellenberg light	.001	.006	.003	.004	.018	.095	.001	.007	.003	.088
Ellenberg moisture	.466	.000	.058	.288	.113	.000	.466	.000	.057	.043
Ellenberg reaction (pH)	.408	.001	.006	.266	.010	.017	.407	.002	.006	.011
Ellenberg nitrogen	.331	.000	.010	.195	.000	.000	.330	.000	.010	.000
Measured soil pH	.288	.014	.005	.137	.013	.020	.287	.013	.006	.043
Measured soil moisture	.448	.000	.012	.298	.004	.001	.447	.000	.012	.039
Measured soil organic	.390	.004	.035	.284	.004	.018	.390	.003	.036	.002

3.4.3. Sciomyzid communities

There were 192 sciomyzid individuals (seven species) collected using pan traps in 2009 (Fig. 3.4) and 19 individuals (three species) in 2010 (Fig. 3.5). *Tetanocera robusta* Loew accounted for 50% (n=96) and 63% (n=12) of the total sciomyzid abundances in 2009 and 2010, respectively. This was followed by *Tetanocera ferruginea* Fallén which represented 43% (n=82) and 32% (n=6) of the sciomyzid catches for 2009 and 2010. In 2009, ca. 11% of variation was attributable to site-specific differences, and ca. 7% of variation could be attributed to habitat status, for pan trap sciomyzid data (Table 3.4), following MRPP analysis. *Tetanocera robusta* was found to be a significant indicator species of wetlands in both 2009 and 2010 (Table 3.6). There were no significant indicator sciomyzid species for infill. A useful NMS ordination could not be constructed from the 2009 sciomyzid data-set due to high variation in abundance data. The 2010 sciomyzid data-set, however, resulted in a 1-dimensional ordination and showed that axis 1 had the strongest correlation (r^2=.205) with vegetation length (Table 3.7).

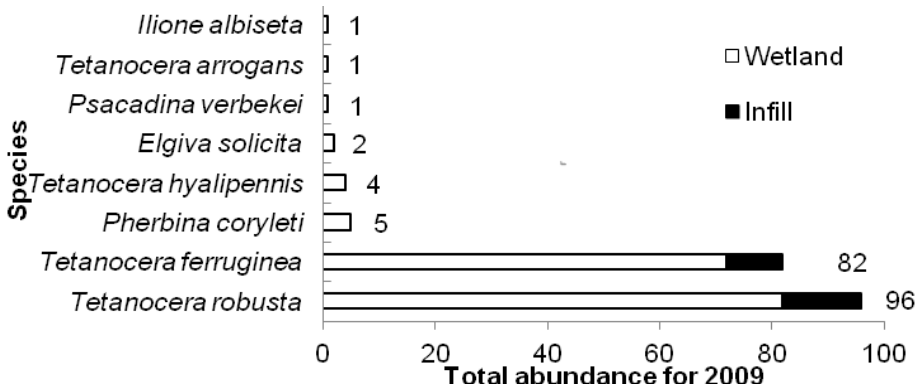

Fig 3.4. *Total sciomyzid species abundances from 2009 (14th July to 13th October) pan trap (n=54) data across all 9 study sites.*

A total of 110 sciomyzids (12 species) were collected by sweep net in 2010 at WG2 with 105 (11 species) and five individuals (three species) collected on the wetland and infill, respectively (Fig. 3.5). Of the 12 species collected in total, *Pherbina coryleti* Scopoli and *Ilione albiseta* Scopoli were the two dominant species, representing 38% and 37% of the total catch, respectively (Fig. 3.6). The dominant species collected using sweep nets (*P. coryleti* and *I. albiseta*) were also different to the dominant species collected with the pan traps (*T. robusta* and *T. ferruginea*) at the same site (WG2). The 2010 pan trap data (Fig. 3.4) for WG2 also showed a higher median abundance of sciomyzids on the wetland (n=5) than the infill (n=0). None of the measured environmental variables showed any significant influence on the sweep net results. For the sweep net sciomyzid data, median abundance of all sciomyzid species was significantly greater ($P<0.01$) on the wetland (n=9.5) than the infill (n=0). Median species richness and median Shannon's entropy were significantly ($P<0.05$) higher on the wetland than the infill, whereas species evenness was higher on the infill (Table 3.3). MRPP on the 2010 sciomyzid data (collected using sweep nets on site WG2), showed that ca. 14% of variation in the data could be attributed to habitat status (Table 3.4). Although MRPP sometimes indicated a low proportion of explained variance, all tests were highly statistically significant for sciomyzids (Table 3.4). Indicator species analysis on sciomyzid data from sweep net samples identified three

species that were indicative of wetlands (Table 3.6): *P. coryleti* (97.6%), *I. albiseta* (83.2%) and *T. ferruginea* (80.2%).

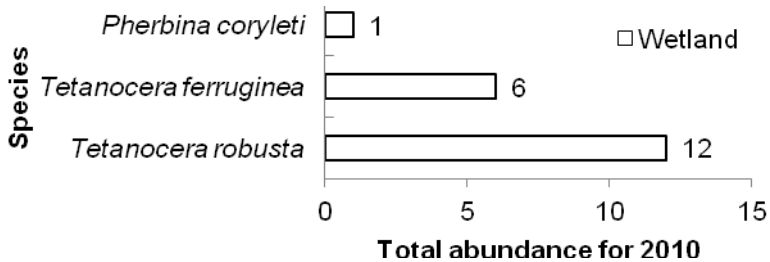

Fig 3.5. *Total sciomyzid species abundances from 2010 (6th May to 30th September) pan traps (n=42) data across 7 study sites.*

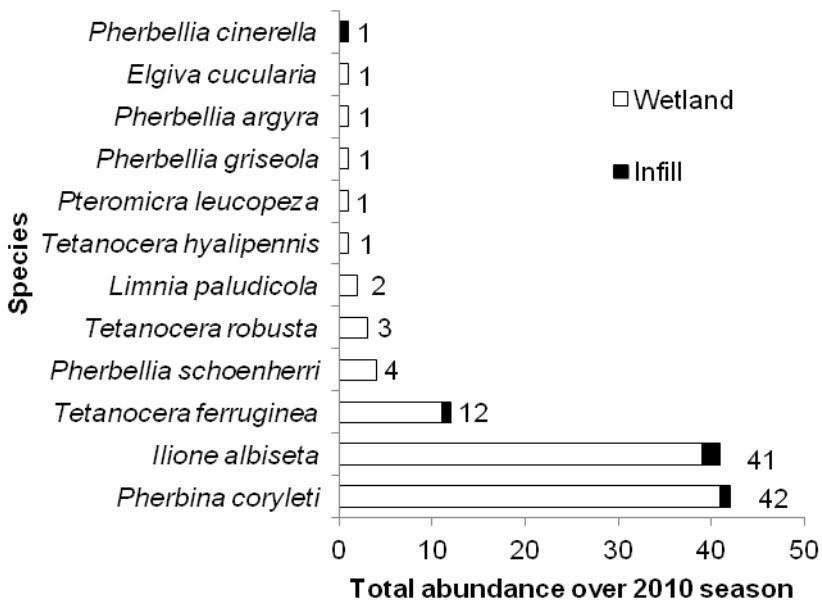

Fig 3.6. *Sciomyzid total abundances (2010 sampling season) for infill and wetland at site WG2 using sweep nets.*

3.5. Discussion

3.5.1. Soils and plant communities

C&D waste infill substrate on wetlands has significantly different properties (pH, moisture and organic content) in comparison to the non-infilled wetlands and this is likely to be responsible for its impact on plant communities as soil properties (including pH, moisture and nitrogen content) are known to be important factors in their composition (Critchley *et al*., 2002; Gough *et al*., 2000; Schultz *et al*., 2011). In addition, substrate disturbance during the process of infilling likely affects plant communities, allowing ruderal species which have the ability to colonise disturbed ground rapidly, to become more dominant (Grime *et al*., 1996). In this study indicator species analysis has proven to be a useful tool in identifying the impacts that the soil properties of C&D waste have on plant communities. *Agrostis stolonifera*, *Cirsium arvense* (L.) Scop, *Festuca rubra* L., *Holcus lanatus* L., *R. repens* and *Trifolium repens*_L. are recognised as species associated with areas of moderate disturbance and soils with a higher (>5) pH (Grime *et al*., 1996). In this study these were, unsurprisingly, among the most significant indicator species of C&D waste infill. *Molinea caerulea* (L.) Moench and *Potentilla erecta* (L.) Rauschel, both described by Grime *et al*. (1996) as being positively associated with low pH soils and undisturbed ground, were significant indicators of acidic wetland areas for this study, having been found extensively on cutover raised bog wetlands. *Cladium mariscus* (L.), although not a significant wetland indicator species in this study, was the dominant species (Table 3.2) in site SW1 (wetland only), and is a species found in wet, neutral to alkaline soils (pH>6), occurring mostly on limestone soils (Conway, 1942), a description befitting this site. As expected there is a strong similarity between the vectors for measured soil properties (pH and percentage moisture content) and estimated values using Ellenberg indices (pH and moisture) in the NMS ordinations (Fig. 3.3 and Table 3.7).

The disposal of C&D waste on wetlands significantly increased the plant species Shannon's entropy, but it is important to note that common ruderal species

accounted for most of this increase. Based on Grime *et al*. (1996), some 57% of all species found on C&D waste infill in this study are known to have a ruderal strategy compared with just 25% of species on the wetlands. Given that the ruderal plant species found in this study are common, their replacing of wetland habitat species, and all the ecosystem services associated with them (Mitsch and Gosselink, 2007), is less than desirable. When all sites were analysed together, 10% (this figure combines the two data from Table 3.4) of the variation in plant communities was attributable to habitat status (infill v wetland). Similarities between infill and wetland are likely due to the presence of bare ground, bryophytes (identification to species level may increase the difference, but this could not be carried out due to time constraints), dead vegetation and a small number of species (*Epilobium hirsutum* L., *Juncus effusus* L. and *Rubus fruticosus* L.) which were found to be present in both C&D infill and wetland quadrats, although not necessarily at the same site, along with the variety of wetland plant communities. This variation can be seen on the NMS ordination (Fig. 3.3a) where the infill plant communities are more clustered than the plant communities of wetland areas. The points for WG1 are likely to be isolated as the dominant species *Elymus repens* (L.) was almost exclusively found on that site. Interestingly, the points for the wet grasslands (wetland area) WG1 and WG2 are at almost opposite corners of the ordination, although they classify as the same habitat under Fossitt (2000). This highlights the limitations of using such a broad habitat classification, which takes abiotic factors into account as well as floristic composition. Following separation of sites according to their pH, MRPP shows that plant communities of acidic wetlands are affected to a greater degree (as shown by higher percentage difference attributed to habitat status) than those of alkaline wetlands by the alkaline C&D waste (Table 3.4). This was expected as there was a greater difference in pH between infill and wetland on acidic sites than alkaline sites. Differences between alkaline and acidic site-specific variation could be explained by the variety of alkaline wetland habitats, compared with acidic wetlands.

3.5.2. Dipteran communities

The impacts of C&D waste on dipteran communities is less clear than the impacts on plant communities (Fig. 3.3) where there is less obvious clustering of sites in the NMS ordination, likely a result of the short distance (10m) between sampling locations. The C&D waste may have had no significant impact on median dipteran (family and morphospecies) richness, Shannon's entropy and evenness on the infill compared to wetland areas. However, there were significant differences (albeit explaining a low proportion of variance) between community compositions according to the habitat status and among sites. This is likely due to the wide variety of ecological associations within many dipteran families (Keiper *et al*., 2002; Oosterbroek, 2006). As shown in Table 3.6, there are a number of dipteran families and morphospecies that were significant bioindicators of both infill (six families and seven morphospecies) and wetland (four families and nine morphospecies). These indicator families changed from generally wetland specialist (Chironomidae, Culicidae), parasitic (Tachinidae), haematophagous (Culicidae) and gall-forming (Cecidomiidae) groups to saprophagous (Anisopodidae, Carnidae, Phoridae, Sepsidae), phytophagous (Chloropidae), haematophagous (Muscidae) and coprophagous (Muscidae, Phoridae) groups, based on published descriptions of the ecology of these families by others (Brake, 2011; Cranston, 1995; McAlpine *et al*., 1981; Oosterbroek, 2006).

The loss of aquatic microhabitats in the infill is the most likely cause for the lower abundances of Chironomidae and Culicidae. The change in plant communities is likely to be the cause of the loss of Cecidomiidae and the gaining of Chloropidae as indicator families after infilling. Although Tachinidae can be found in many habitat types, many species may be habitat-specific due to their host species (phytophagous insects such as Lepidoptera) specificity (Stireman III *et al*., 2006). As a result, their occurrence on the infilled sites may be limited. The family Chironomidae was found to be a significant indicator of wetlands, a finding that is supported elsewhere (Cranston, 1995). The MRPP results also suggest that infilling with C&D waste may be more detrimental to acidic wetland dipteran communities than those on an alkaline wetland, as slightly higher proportion of

variation in dipteran composition can be attributed to habitat status (Table 3.4) for the dipteran communities of the acidic sites (6%) than for alkaline sites (4%) regardless of site-specific differences being higher in the latter (32% in alkaline sites versus 14% in acidic sites).

3.5.3. Sciomyzid communities

Fifty nine species of Sciomyzidae are currently known in Ireland (Speight and Knutson, 2012), 14 of which were collected during this study. The C&D waste appears to have had a significant impact on sciomyzid communities. Pan trap data showed significant (P=0.005) differences between the sciomyzid communities of the infill and wetland areas even though the distance between the pan traps of each area was only ten metres. *Tetanocera robusta* was a significant (P<0.05) indicator species for wetlands in both years. Sweep net sampling also showed a significant (P=0.005) difference between infill and wetland, with *P. coryleti* being a significant (P<0.005) indicator species for wetland. These findings support previous studies showing that sciomyzids display limited movement and are habitat specific (Speight, 2004; Vala & Brunel, 1987; Williams *et al*., 2009, 2010).

The most dominant species caught using the pan traps (for 2009 and 2010) was *T. robusta*, followed by *T. ferruginea*. The dominance of *I. albiseta* and *P. coryleti* in the sweep net data when compared with pan traps, supports previous findings (Williams, 2007), and indicates their limited movement. At site WG2 sweep nets and pan traps gave very different results. Pan traps collected only *T. robusta* and *T. ferruginea*, whereas the sweep nets collected 12 species. This difference between trapping methods should be considered if using sciomyzids as bioindicators. The significant wetland indicator sciomyzid species (*T. robusta* and *P. coryleti*) are known to prey and feed on multiple aquatic snail species from several genera as larvae (Speight and Knutson, 2012) as are all sciomyzids found on C&D waste (with the exception of *Pherbellia cinerella* Fallén which preys upon a range of terrestrial and semi aquatic gastropods). Interestingly, although these aquatic snails usually prefer alkaline water and the C&D waste is more

alkaline than the wetlands, there was a decrease in the abundance of sciomyzids collected on the C&D waste. This was likely due to a shortage of appropriate aquatic microhabitats on the C&D waste for the aquatic snails and sciomyzid larvae.

Pherbellia cinerella was the only species to have been collected (one individual with sweep net) solely on the C&D waste infill. This was not surprising given its known occurrence on dry habitats (Speight and Knutson, 2012). All of the other sciomyzid species collected are associated almost exclusively with 'wet' habitats (Speight and Knutson, 2012). The observation here that these species were all either exclusively found on wetlands, or had a large majority on the wetlands, highlights their sedentary nature and usefulness as bioindicators of wetland habitat change. The low number of sciomyzid individuals collected is a trait of the family which is not unknown, although there are some species (such as *I. albiseta*) that are often found in high numbers (Williams *et al*., 2009; Williams *et al*., 2010).

3.5.4. Problems encountered

From the start of this study, it was found that wetlands were often perceived by landowners as valueless land, especially if peat had already been cut away from bogs, or if they were too wet to graze with livestock. Landowners sought to improve the land by covering the infill with topsoil to produce agriculturally productive grasslands. Permits specifically for the disposal of C&D waste have been available from local authorities in Ireland since 2001.

The almost complete absence of published information on ecological impacts of infilling wetlands with C&D waste is not aided by the problems associated with studying these sites. Permission to undertake ecological work has to be obtained from the landowner and this permission is likely to be refused at any time given the nature of the activity, as happened on site SW2 at the start of year two. The disturbance of sites by heavy machinery and the process of infilling had been anticipated as a potential problem at the beginning of the study, but this is very difficult to predict as sites that appear 'dormant' can become active at any time (as happened with site CB5 at the beginning of year two), depending on the volume

of waste being produced in the locality. Two more otherwise suitable sites had to be removed from the study at the beginning of Year 1 (in addition to the loss of individual samples from several sites over the study) as a result of human interference with equipment being stolen and broken repeatedly. As these (usually poorly fenced) sites were frequently located near rural-urban interfaces and major roads, they were highly visible to members of the public. Shortly after our studies commenced at some of the sites, sampling was compromised by members of the public moving or emptying traps and/or flattening. There were also unforeseen restrictions regarding invertebrate sampling, in particular, the limitations of using sweep nets caused by the hazardous topography of infilled sites.

Currently, some inspections of waste composition at C&D waste sites are carried out by the local authority but daily inspections of waste are the responsibility of the waste haulier and landowner. Although the waste permits were almost always granted to landowners with the condition of using the land for agricultural purposes afterwards, it was frequently found in this study (particularly for sites owned by building developers) that these sites were left without topsoil or had been further developed (residential or commercial buildings or yards). There is a possibility that this type of permit could provide a route for such surreptitious development of wetland areas, for which it would be difficult to get permission directly.

3.5.5. Recommendations

Our results indicate that the infilling of wetlands in this study with C&D waste has resulted primarily in the replacement of wetland plant communities with ruderal plant species and a reduction in wetland specialist Diptera. Given these dramatic changes, it is likely that the wetland ecosystem function of the sites studied has been impaired. However, the degree and significance of impairment will depend on the resilience of the wetland which in turn depends, *inter alia*, on the wetland type, the size of the wetland, its connectivity with other wetlands and the proportion of the wetland infilled with C&D waste. While it could be argued that any loss in wetland ecosystem function should be avoided at all costs, the

reality is that, in the absence of complete C&D waste recycling, wetlands, particularly those not included in Natura 2000, will continue to be infilled for the foreseeable future. With this in mind, we would make the following recommendations:

- All future waste permits (regardless of site size or C&D tonnage) should require either an independent Environmental Impact Assessment or an Appropriate Assessment. These assessments should include, as a minimum, surveys of plants, wetland invertebrate assemblages and wetland vertebrates in addition to the collection of physical data on soils and hydrology. If permission is granted, these surveys will provide baseline data for future monitoring of the sites.

- Annual ecological surveys should be undertaken by local authorities or a third party authorised by them, to monitor changes in the wetlands after infilling has commenced (to determine if impacts change with time) and to ensure that licensing agreements are adhered to.

- Invertebrate baseline monitoring stations, protected by security fencing to avoid damage by vandalism, should be set up (prior to and during infilling). Once these secure stations are in place, more visible invertebrate sampling methods such as malaise, emergence and pitfall traps could be employed.

- The above increased ecological monitoring could contribute to a database used to inform decisions regarding appropriate site selection for C&D infilling, thereby preserving those wetlands which are most vulnerable.

- At least part of the wetland sites used for C&D infill should be protected from future infilling activities. The initial ecological assessment could be used to determine the most ecologically valuable area to protect.

3.6. Conclusions

The infilling of wetlands in this study with C&D waste has had an impact on soil properties and plant communities. Dipteran communities were also affected by the C&D waste infill, probably as a result of the changes in plant communities and the loss of 'wet' areas. There are many potential problems with carrying out such studies and recommendations have been given to overcome these. Given the paucity of research in this area, this study highlights that the infilling of wetlands with C&D waste can have serious consequences for wetland ecology.

Future research should focus on C&D waste infill sites of different waste composition and age, and in areas of different geographical, geological, topographical and meteorological settings. Following further research, this information could be used by planning authorities to aid in future policy making and in the development of sustainable C&D waste management strategies.

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Chapter 4:

Assessing metal contamination from construction and demolition (C&D) waste used to infill wetlands: using *Deroceras reticulatum* (Mollusca: Gastropoda).

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4. Assessing metal contamination from construction and demolition (C&D) waste used to infill wetlands: using *Deroceras reticulatum* (Mollusca: Gastropoda).

4.1. Abstract

Large quantities of construction and demolition waste (C&D) are produced globally every year, with little known about potential environmental impacts. In the present study, the slug, *Deroceras reticulatum* (Mollusca: Gastropoda) was used as the first biomonitor of metals (Ag, As, Ba, Cd, Co, Cr, Cr, Cu, Mn, Mo, Ni, Pb, Sb, Se, Ti, Tl, V and Zn) on wetlands post infilling with construction and demolition (C&D) waste. The bioaccumulation of As, Ba, Cd, Co, Sb, Se and Tl were found to be significantly elevated in slugs collected on C&D waste when compared to unimproved pastures (control sites), while Mo, Se and Sr had significantly higher concentrations in slugs collected on C&D waste when compared to known contaminated sites (mining locations), indicating the potential hazardous nature of C&D waste. Identifying precise sources for these metals within the waste can be problematic, due to its heterogenic nature. Biomonitors are a useful tool for future monitoring and impact studies, facilitating policy makers and regulations in other countries regarding C&D waste infill. In addition, improving separation of C&D waste to allow increased reuse and recycling is likely to be effective in reducing the volume of waste being used as infill, subsequently decreasing potential metal contamination.

4.2. Introduction

Wetlands are among the world's most important habitats, providing many ecologically and economically important ecosystem services including water storage and filtration, flood control, carbon fixation, and habitat provision (Mitsch and Gosselink, 2007; Keddy, 2000). Covering an estimated nine million km^2 globally, they include habitats such as swamps, bogs, fens, marshes and wet grasslands which occur from polar to tropical latitudes (Keddy, 2000). Despite their importance, many wetlands have been and continue to be significantly impacted by anthropogenic activities, including draining, dredging and infilling (Mitsch and Gosselink, 2007). While draining is responsible for the largest amount of wetland loss, infilling is also a significant contributor (Mitsch and Gosselink, 2007), with construction and demolition (C&D) waste often being used under license (Council Directive 2006/12/EC; Statutory Instrument No. 165/1998; Statutory Instrument No. 821 of 2007) for this purpose (EPA, 2012).

Construction and demolition waste results from the construction, renovation or demolition of any structures, such as buildings, roads and bridges (Franklin Associates, 1998; Clarke *et al*., 2006). For example C&D wastes produced on building sites is dependent on factors such as variations in regional building practices, such as the increased use of timber in Scandinavian countries (European Commission DG ENV, 2011), and the structure, size and nature of source activity (Franklin Associates, 1998; Weber *et al*., 2002). The contents of C&D waste are therefore variable and can include materials such as soil, stones, concrete, timber, plastics, gypsum, metal and bitumen (Franklin Associates, 1998; Williams, 1998), some of which may contain potentially hazardous metals (e.g. Cu, As, Pb, Cd) and other environmentally important compounds such as benzene and chromates (Weber *et al*., 2002). Globally, large quantities of this waste are generated on an annual basis with production linked to economic growth (Eurostst, 2011; Staunton *et al.*, in prep.). The most recent European Union data suggest over 870 million tonnes of C&D waste was produced in 2008 (Eurostat, 2011). However, this data may be unreliable, as weight/volume estimation techniques are open to biased reporting, and even among countries different materials are reported as C&D

waste (European Commission DG ENV, 2011). The rates of production in many eastern European countries are also known to be under-reported (Eurostat, 2011) and significant amounts of unregulated C&D waste disposal are known to occur in Spain, Hungary (European Commission DG ENV, 2011), Italy (Symonds Group *et al*., 1999) and the United States (ICF Incorporated, 1995). Most of the waste is disposed of in unlined landfills (Franklin Associates, 1998; Williams, 1998), but there is no published information on the habitat types these landfills affect, or the areas covered. Although all European wetlands designated as Natura 2000 sites are protected under the Habitats Directive (Metcalfe-Smith *et al*., 1996), local authorities in Ireland issue permits for infilling of undesignated wetlands with C&D waste, and only a small number of these applications require the completion of an Environmental Impact Assessment (EIA) (Council Directive 2006/12/EC; Statutory Instrument No. 165/1998; Statutory Instrument No. 821 of 2007).

Although any hazardous material should be removed from the waste prior to infilling in unlined landfills, some of it inevitably fails to be adequately removed during the sorting process (Roussat *et al*., 2008). Leachate from C&D waste can contain elevated levels of metals including Al, Fe and Mn (Weber *et al*., 2002; Melendez, 1996), and priority pollutants (USEPA, 2014) such as As, Cd, Cu and Pb (Weber *et al*., 2002; Roussat *et al*., 2008; Melendez, 1996; López and Lobo, 2014). These elevated metal concentrations in C&D waste leachate can pose a risk to human health if they enter water supplies (Roussat, 2008; Melendez, 1996). The generation of this leachate occurs as surface and groundwaters move through the waste, mobilising both organic and inorganic compounds (Faeiza *et al*., 2004). However, leachate pollutant concentrations can vary according to waste permeability and depth, age of the waste and exposure time. As the C&D waste is typically heterogeneous in nature, there are logistical difficulties in obtaining representative samples for analysis of contaminant content. In addition, the evaluation of temporal variations in contaminant concentrations is restricted by sampling leachate at one point in time. Furthermore, the direct chemical analysis of the waste or the waste leachate limits the provision of information on contaminant bioavailability and ultimately potential toxicity (Hu *et al*., 2012; Leita *et al*., 2013; Mowat and Bundy, 2001).

The use of biomonitors is a well-established technique for monitoring bioavailable levels of environmental contaminants in terrestrial (Greville and Morgan, 1991; Boshoff *et al*., 2013; Popham and D'Auria, 1980) and aquatic (Pyatt *et al*., 1997; Niyogi *et al*., 2014) ecosystems. There is little published information on the ecological effects of C&D waste disposal in wetlands globally except one recent study (from Ireland) (Staunton *et al*., 2014; See Chapter 3) which showed that infilling of wetlands with C&D waste significantly altered the plant and dipteran communities present. However, to date organisms have never been employed for monitoring potentially toxic metal contamination from C&D waste. Terrestrial molluscs (Berger and Dallinger, 1993; Metcalfe-Smith *et al*., 1996), in particular slugs (Bullock *et al*., 1992; Triebskorn and Köhler, 1996), have been shown to bioaccumulate metals and have been used as cost effective (Marigomez *et al*., 1998)) biomonitors of metals at locations contaminated as a result of mining activities (Greville and Morgan, 1989a; Greville and Morgan, 1990). *Deroceras reticulatum* (Müller, 1774) is found extensively on wetlands infilled with C&D waste in Ireland. This slug fulfils the prerequisites considered to be essential for a useful biomonitor (Jones and Kaly, 1996), including; being geographically widespread (on a global scale), possessing an annual life cycle (adults dying upon the first winter frosts) (Wiktor, 1999), limited active dispersal ability (Aubrey *et al*., 2006; Baur, 1993), easily collectable (Bullock *et al*., 1992; Greville and Morgan, 1989a; Greville and Morgan, 1990) and identifiable (Wiktor, 1999), and amenable to laboratory studies (Triebskorn and Köhler, 1996). While metal uptake can occur from soil by absorption through the digestive tract or through the dermis (Peijnenburg, 2002), molluscs tend to accumulate the majority of metals from ingested food (Peijnenburg, 2002; Gräff *et al*., 1997; Dallinger *et al*., 1997; Notten *et al*., 2005; Croteau and Luoma, 2008) with their tissue metal content being indicative of ambient plant and soil metal concentrations (Notten *et al*., 2005).

The primary aim of this study was to assess the environmental impact of infilling wetlands with C&D waste, in terms of metal bioavailability by employing for the first time *D. reticulatum* as a biomonitor of metal contamination on the waste.

4.3. Methods

4.3.1. Study area

Slug samples were collected from nine sites in Ireland (see details in Fig. 4.1 and Table 4.1) which consisted of three C&D waste sites (CD), three known contaminated sites (KC), and three sites which were considered pristine (PR). These sites were selected on the basis of representing different levels of metal contamination which should be reflected in the metal content of the slug tissue. *Deroceras reticulatum* was present on all nine sites and the presence of short vegetation permitted the use of slug refuge traps (see below). The CD sites which are typical of C&D waste infill sites throughout Ireland and indeed Europe, included C&D waste on wet grassland (CD1), reed and large sedge swamps (CD2) and peatland (CD3). All three sites were licensed after 2001 (and infilled before 2009), and contained, for the most part, concrete, bitumen, soil and stone. CD2 was still being actively infilled at the time of this study, but none of the sites had been levelled or covered with topsoil. The PR sites, located in rural areas, > 5km from municipal and industrial centres were pastures where no chemical treatments including, fertilisers and pesticides had been applied for at least 50 years and hence were considered pristine and selected as controls for comparative purposes.

Table 4.1. *Categorisation of study sites*.

Category	Description	Label
C&D waste	C&D waste on wetland	CD1
C&D waste	C&D waste on wetland	CD2
C&D waste	C&D waste on wetland	CD3
Considered pristine	Unimproved pasture	PR1
Considered pristine	Unimproved pasture	PR2
Considered pristine	Unimproved pasture	PR3
Known contaminated	Closed mine (Silvermines – Magcobar)	KC1
Known contaminated	Closed mine (Silvermines – Shallee)	KC2
Known contaminated	Closed mine (Tynagh)	KC3

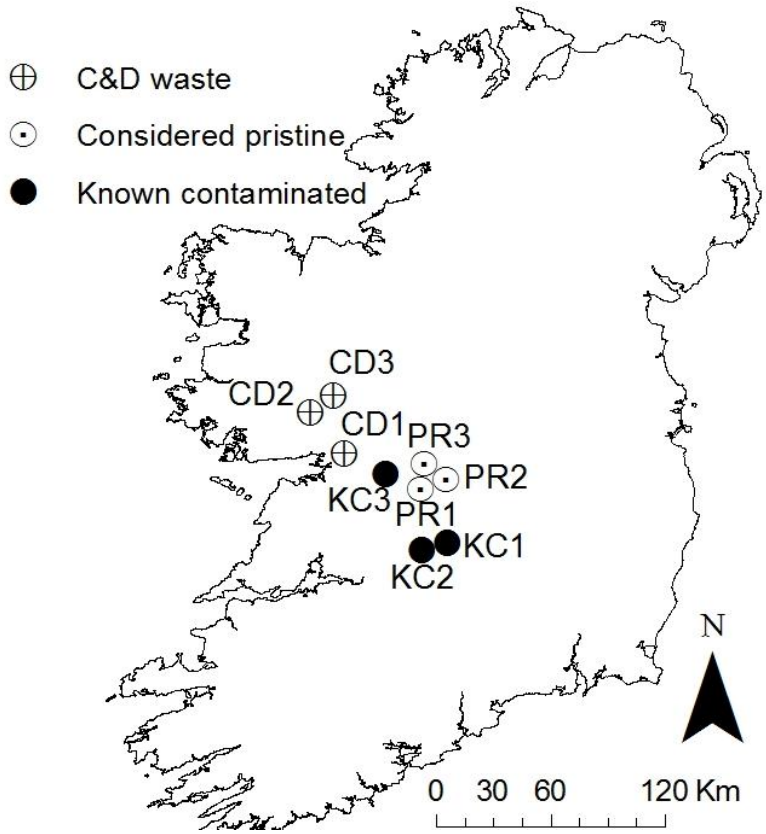

Fig. 4.1 *Site locations in Ireland. CD = construction and demolition waste; PR = considered pristine (unimproved pasture); KC = known contaminated (mine).*

4.3.2. Experimental procedure

The variability in metal concentrations for small samples of slugs has been documented (Gräff *et al*., 1997) for Cd, Pb and Zn. Standard deviations often larger than the mean concentrations have been recorded with sample sizes of only three specimens (Gräff *et al*., 1997). Relatively smaller standard deviations were found in other studies (Greville and Morgan, 1989a; Greville and Morgan, 1990; Greville and Morgan, 1991; Greville and Morgan, 1989b; Greville and Morgan, 1993) with sample sizes of up to 18. To address the limitations of previous studies, a larger sample size (n = 30) and parametric range (18 elements) were used in the present study. At each site adult *D. reticulatum* (n = 30) were collected over 2 days in September, 2011 using 36 (60 x 60 cm) refuge traps placed 2 m apart in a 6 x 6 grid. Samples were transported to the laboratory in clean

polythene bags (one slug per bag; at 4 °C during transportation) and rinsed using Milli-Q (Millipore, Bedford, USA) water. Depuration was allowed (48 hours at 4 °C) in clean plastic containers (1 slug per container) using damp filter paper (changed after 24 hours to minimise coprophagy). The slugs were further rinsed with Milli-Q and freeze dried (Freezone 12, Labconco, Kansas City, USA) at -50 °C. Sample decomposition was performed using a microwave sample preparation system (Multiwave 3000, Anton Paar, Graz, Austria). Samples were digested in a class 10,000 (ISO class 7) clean room using 4 cm^3 of HNO_3 (Trace Metal Grade, 67-69%, Fisher, UK) and 2 cm^3 of H_2O_2 (*Trace*SELECT® Ultra ≥30%, SIGMA-ALDRICH, USA). Metal concentration (Ag, As, Ba, Cd, Co, Cr, Cu, Mn, Mo, Ni, Pb, Sb, Se, Sr, Ti, Tl, V, Zn) was determined using Inductively Coupled Plasma Mass Spectrometry (ICP-MS; ELAN DRC-e, Perkin Elmer, Waltham, USA) in a class 1000 (ISO class 6) clean room. Certified Reference Materials (CRM) of TORT-2 (lobster hepatopancreas; National Research Council Canada) and NIES No.6 (*Mytilus edulis*; National Institute for Environmental Studies Japan) were used with method blanks to validate the accuracy of data for quality assurance purposes.

4.3.3. Statistical analysis

The Anderson-Darling test was used to test for data normality. The Kruskal-Wallis test with a Dunn's multiple comparison post-hoc test was used to determine where significant differences in slug metal concentration occurred among site categories. All statistical calculations ($P < 0.05$ and $P < 0.01$) were performed using Minitab (version 16) and SPSS (version 20). SigmaPlot (version 12.0) was used to create graphs.

4.4. Results and Discussion

All previous studies employing *D. reticulatum* as a biomonitor of environmental contamination have focused on Cd, Cu, Pb and Zn (Greville and Morgan, 1989a; 1989b; 1990; 1991; 1993). In addition to these EU-List Priority Substances (Council Directive 2008/105/EC; Council Directive 2006/11/EC) the present study included a further 13 metals (Ag, As, Ba, Co, Cr, Cr, Mo, Ni, Sb, Se, Ti, Tl, V) considered a significant risk to environmental quality and included in the EU-List II Priority Substances (Mn was also included, though not a Priority Substance). At elevated concentrations, all 18 elements are potentially toxic (Adriano, 2001; Rainbow, 1997; Kuperman *et al*., 2004; 2006).

The results from the analysis of the *Mytilus edulis* and lobster hepatopancreas reference tissues (Table 4.2) are in good agreement with their respective certified ranges. These reference materials represent the closest possible matrix match for slug tissue which is currently commercially available. Metal concentrations are similar to what was potentially expected in the present study, based on previous investigations on the metal content of *D. reticulatum* tissue from a range of sites (Greville and Morgan, 1989a; 1989b; 1990).

Metal concentrations in *D. reticulatum* are presented for each site category (mean value from three sites in each category) in Table 4.3, while Figs 4.2, 4.3 and 4.4 show individual site data. Zinc exhibited the highest median concentration of all elements measured across all C&D waste site samples (207.83 µg g^{-1}), while Mn had the highest median value across all mine (731 µg g^{-1}) and unimproved pasture site samples (226.3 µg g^{-1}). As micronutrients, both Mn and Zn (USEPA, 2001) are broadly more abundant in biological tissue when compared to other metals ()Laskowski and Hopkin, 1996). Kruskal-Wallis tests performed on the data showed that all 18 metals displayed some significant ($P < 0.05$) differences between samples recovered from the different site categories (Table 4.4). There were 11 significant ($P < 0.05$) differences for data comparisons between C&D waste and pasture samples, while comparisons of both 'C&D waste vs mining' and 'mining vs pasture' samples each showed 16 significant ($P < 0.05$)

differences.

Table 4.2. *Observed results from analysis of Certified Reference Materials, with certified and reference values. All values are µg g^{-1}.*

Element	TORT-2 certified value (± SD)	Observed this study (± SD)
Ag	-	-
As	21.6 (± 1.8)	21.53 (± 1.02)
Cd	26.7 (± 0.6)	27.82 (± 1.24)
Co	0.51 (± 0.09)	0.47 (± 0.05)
Cr	0.77 (± 0.15)	1.27 (± 0.37)
Cu	106 (± 10)	109.82 (± 9.63)
Mn	13.6 (± 1.2)	13.82 (± 2.03)
Mo	0.95 (± 0.10)	0.98 (± 0.05)
Ni	2.50 (± 0.19)	2.30 (± 0.31)
Pb	0.35 (± 0.13)	0.34 (± 0.08)
Se	5.63 (± 0.67)	6.09 (± 0.57)
Sr	45.2 (± 1.9)	52.41 (± 7.51)
Zn	180 (± 10)	180.30 (± 10.27)
Element	NIES no.6 certified value (±SD)	Observed this study (± SD)
Ag	0.027 (± 0.003)	0.044 (± 0.040)
As	9.2 (± 0.5)	9.56 (± 0.56)
Cd	0.82 (± 0.03)	0.85 (± 0.03)
Co	0.37 (reference value)	0.30 (± 0.04)
Cr	0.63 (± 0.07)	1.08 (± 0.31)
Cu	4.90 (± 0.30)	6.18 (± 1.05)
Mn	16.3 (± 1.2)	15.19 (± 1.44)
Mo	0.95 (± 0.10)	0.85 (± 0.05)
Ni	0.93 (± 0.06)	0.80 (± 0.10)
Pb	0.91 (± 0.04)	0.81 (± 0.12)
Se	1.50 (reference value)	1.65 (± 0.57)
Sr	17.00 (reference value)	17.43 (± 1.30)
Zn	106 (± 6)	106.48 (± 7.74)

Concentrations of As, Ba, Cd, Co, Sb, Se and Tl were significantly ($P < 0.05$) elevated in slugs from C&D waste sites when compared to unimproved pasture (Table 4.4). Arsenic and Cd (Weber *et al*., 2002; Melendez, 1996) have been reported at elevated concentrations in C&D leachate, from simulations used in the laboratory, small field test cells and full scale C&D waste infill sites (Weber *et al*., 2002; Melendez, 1996; López and Lobo, 2014). Isolating the source(s) of the increased metal concentrations within the C&D waste in this study is difficult due to its variable nature. They are generally known to originate from permitted

substances such as pigments used on C&D waste materials (for Cd and Sb), wood treated with preservatives (As), and cement (Tl) (Weber *et al*., 2002; Adriano, 2001). Another likely source of these metals is from unpermitted items or substances mixed through the C&D waste, some of which may be hazardous, such as municipal waste (As, Cd), electrical equipment (Cd, Sb, Tl) and pesticide / paint containers (As, Cd, Sb) (Weber *et al*., 2002; Adriano, 2001; Roussat *et al*., 2002). Determining the abundance of these unpermitted items present in C&D waste is not feasible due to the volume and associated cost, but the presence of such items was noted frequently at the C&D waste sites.

Concentrations of three essential (Adriano, 2001) metals (Mo, Se, Sr) were significantly higher ($P < 0.05$) in the slugs from C&D waste sites than from mines (KC1, 2 and 3). A nationwide soil geochemical atlas (Fay *et al*., 2007) suggests that the C&D waste sites have higher background levels for Mo, Se and Sr, compared to KC1 and KC2, and this is reflected in the slug tissue concentrations. Soil S (in the form of sulphate) and P compete for the same uptake pathways as Mo in plants (Heuwinkel, 1998; Alhendawi, 2005), and as a result, uptake of Mo by plants may be reduced in mining locations with higher concentrations of S and P. Likewise a correspondingly lower concentration of Mo would, therefore be expected in herbivorous slugs such as *D. reticulatum* on these sites. A similar scenario could be expected for Se which is known to share similar uptake pathways in plants as S (Sors *et al*., 2005). The significantly higher Sr concentrations in slugs collected on C&D waste (compared to mines) is likely to be a result of elevated background soil Sr concentrations at these C&D waste locations (Fay *et al*., 2007). All of the metals which showed significantly elevated levels in the slug samples from C&D waste (compared to mines and unimproved pasture) are EU priority contaminants (Council Directive 2008/105/EC; Council Directive 2006/11/EC).

Table 4.3. *Median metal concentrations in Deroceras reticulatum for each site category (n = 90 per site category). Interquartile range in parentheses. All values are µg g^{-1}.*

Element	Mining	C&D	Pasture
Ag	0.454 (0.846)	0.000 (0.015)	0.036 (0.045)
As	0.378 (1.046)	0.1864 (0.0997)	0.1284 (0.0622)
Ba	78.9 (70.7)	9.937 (5.824)	2.672 (4.496)
Cd	37.20 (29.06)	6.570 (4.692)	3.873 (3.530)
Co	0.6114 (0.6609)	0.3851 (0.2737)	0.3053 (0.1627)
Cr	1.024 (0.416)	0.9027 (0.2924)	0.9454 (0.4484)
Cu	103.59 (52.21)	46.03 (18.34)	49.99 (24.73)
Mn	731 (803)	97.4 (119.6)	226.3 (382.8)
Mo	1.817 (0.909)	3.079 (1.279)	2.835 (2.632)
Ni	1.765 (1.512)	1.0570 (0.5185)	1.272 (0.931)
Pb	7.3 (59.8)	0.3099 (0.4105)	0.3385 (0.2383)
Sb	0.0545 (0.1073)	0.0154 (0.0225)	0.0095 (0.0154)
Se	0.7040 (0.7367)	1.8684 (1.4203)	0.7062 (0.6192)
Sr	43.43 (21.22)	48.83 (38.47)	54.22 (28.95)
Ti	38.600 (6.838)	31.265 (5.582)	38.253 (10.796)
Tl	0.434 (0.940)	0.0167 (0.0121)	0.0115 (0.0077)
V	0.1014 (0.0751)	0.1223 (0.0541)	0.1286 (0.0509)
Zn	795.3 (519.3)	207.83 (70.01)	197.31 (96.14)

Overall metal concentrations in slugs were generally found to be significantly ($P < 0.05$) lower from both C&D waste (Ag, As, Ba, Cd, Co, Cu, Mn, Ni, Pb, Sb, Ti, Tl, Zn) and unimproved pasture (Ag, As, Ba, Cd, Co, Cr, Cu, Mn, Ni, Pb, Sb, Tl, Zn) when compared to mine sites. In particular, concentrations of Tl, Sb, Mn, Zn, Ba, Cd, Cu and Pb were elevated in *D. reticulatum* from the mine sites (Figs 4.2, 4.3 and 4.4). Environmental contamination associated with past mining operations at these locations have been widely documented (Stanley *et al*., 2009; Brogan, 2003; Office of Environmental Enforcement, 2004; Henry, 2014). The significantly lower concentrations in *D. reticulatum* from the C&D waste and pasture sites, compared with the mine sites coincided with metals that were known to exist at elevated concentrations in soils at KC3 – Tynagh Mines (Stanley *et al*., 2009; Brogan, 2003) (As, Ba, Cd, Cu, Ni, Pb, Zn) and Silvermines KC1 and 2 (Stanley *et al*., 2009; Office of Environmental Enforcement, 2004) (Cd, Pb, Zn). All slug samples were collected on vegetated mine tailings, with KC2 having a disused smelting plant and laboratories adjacent to the sampling location.

Table 4.4. *Comparison (between site categories; n = 90 per site category) of metal concentrations in Deroceras reticulatum, using Kruskal-Wallis test with Dunn's multiple comparison post hoc test. Values shown are test statistic 'K'; * indicates significant difference at P ≤ 0.05; ** indicates significant difference at P ≤ 0.01.*

Between sites	Mining vs Pasture	Mining vs C&D	C&D vs Pasture
Ag	100.52**	160.43**	59.92**
As	116.03**	72.38**	-42.65**
Ba	166.55**	101.52**	-65.03**
Cd	156.26**	109.21**	-47.04**
Co	84.86**	53.96**	-30.91*
Cr	15.24*	30.44	15.21
Cu	120.84**	127.56**	6.72
Mn	75.31**	136.16**	60.84**
Mo	-63.08**	-87.95**	-24.87
Ni	58.43**	95.01**	36.58**
Pb	134.27**	135.23**	0.96
Sb	118.07**	87.47**	-30.60**
Se	-2.03	-89.17**	-87.13**
Sr	-56.02**	-39.55**	16.47
Ti	-10.94	91.91**	102.86**
Tl	153.42**	114.55**	-38.87**
V	-44.63**	-26.72	17.91
Zn	142.13**	127.53**	-14.6

Compared to C&D waste, only four metals (Ag, Mn, Ni, and Ti) had significantly ($P < 0.05$) higher concentrations in slugs collected on pasture sites (Table 4.3), with two of these (Mn and Ti) significantly ($P < 0.05$) elevated in specimens from one site (PR2) in particular (compared to PR1 and PR3; Fig 4.2 & 4.3). In addition Mo, Sr and V also displayed significantly higher concentrations in slugs from the unimproved pasture than from mining sites. Previous studies with the aquatic snail *Lymnaea stagnalis* (Linneus L.) and the slug *Arion ater* (Linnaeus, 1758) have similarly shown control samples (unpolluted canal and remote hilltops respectively) to have elevated concentrations of Mn, Sr and Ti (Popham and D'Auria, 1980; Pyatt *et al*., 1997), with no known reasons for the increased metal concentrations. Soil geochemical profiles (Fay *et al*., 2007) show background Mn, Ni and Ti levels to be similar for the C&D waste and unimproved pasture sites, while Mo, Sr and V are thought to be higher around the pasture sites than KC1 and KC2, although some localised variability is possible. In addition, Mn can

occur naturally at elevated concentrations in limestone (Homoncik *et al*., 2010), which is the dominant lithology at all of the study sites. The uptake of Sr is thought to be reduced in *A. ater* with increasing concentrations of Pb (Popham and D'Auria, 1997), so the low levels of Pb on the pasture (compared to mines; Table 4.4) may contribute to more efficient Sr accumulation. Nickel is thought to share a common poorly regulated uptake pathway with Co in the aquatic snail, *L. stagnalis* (Niyogi *et al*., 2014), so the elevated concentrations of Co at the C&D waste (Table 4.4) sites may compete directly with Ni, thereby reducing uptake of the latter.

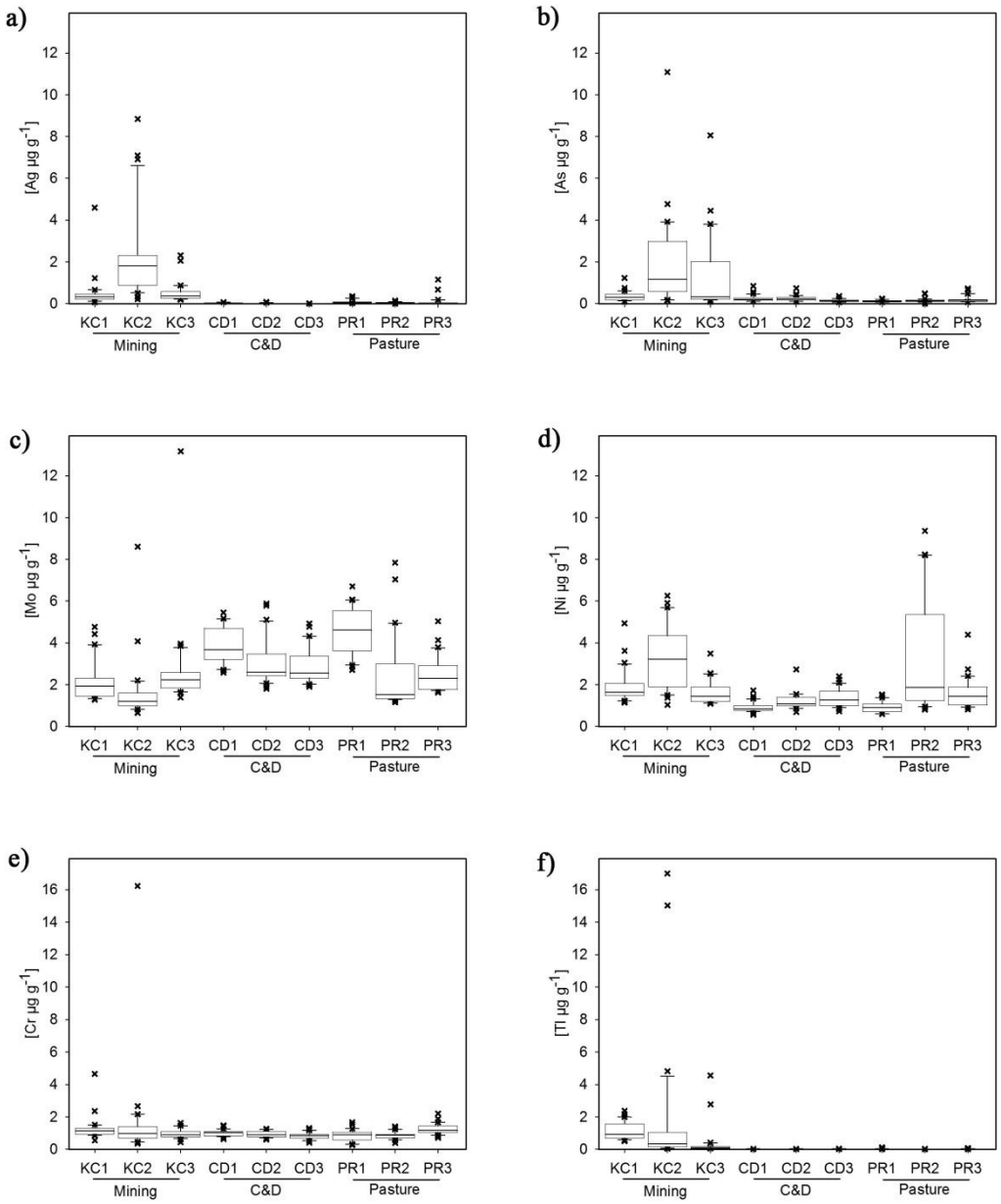

Fig. 4.2 *(a-f) Boxplots showing median metal concentrations and outliers for Ag, As, Mo, Ni, Cr and Tl (data separated by site and sites grouped by category; n = 30 for each site).*

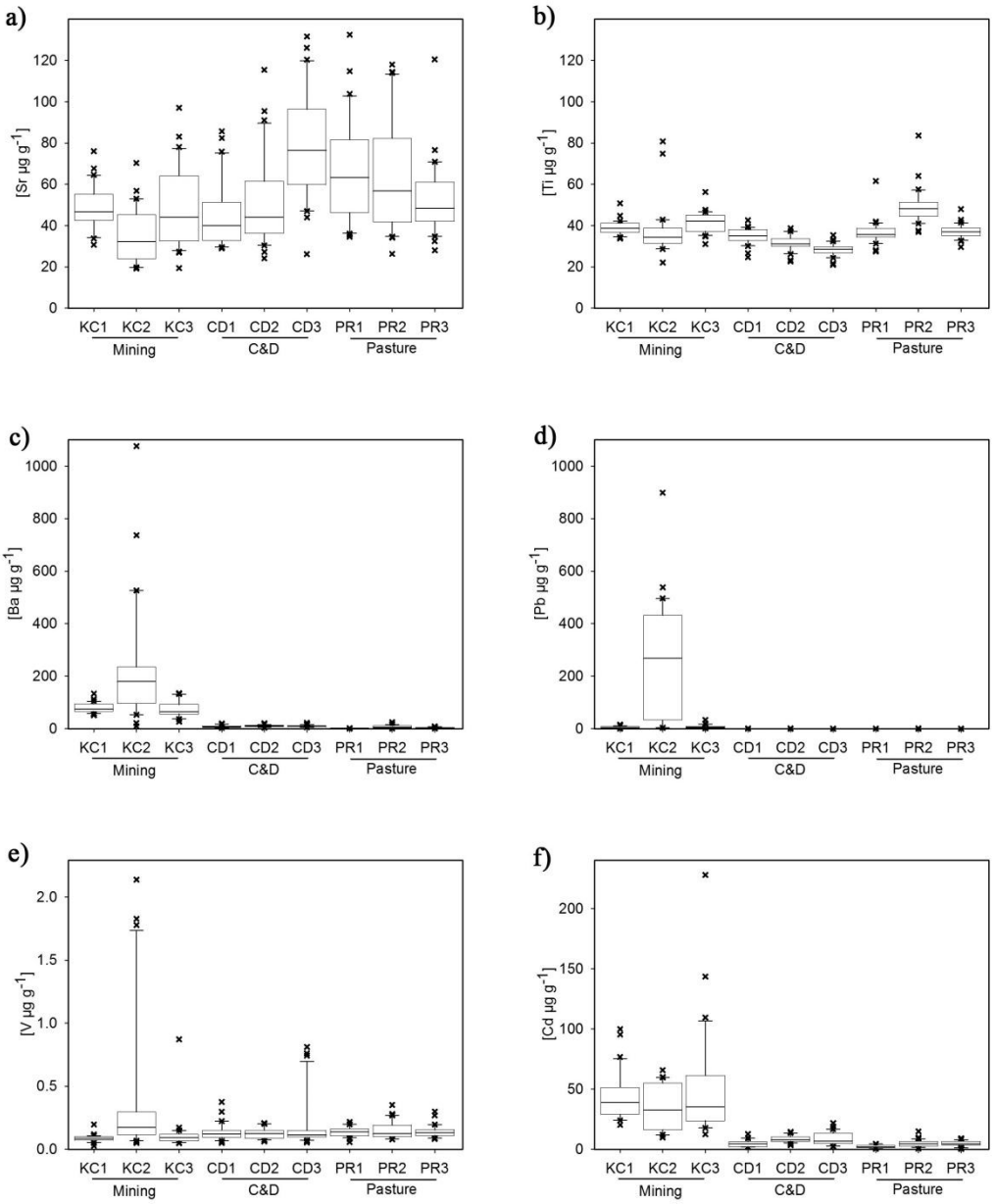

Fig. 4.3 *(a-f) Boxplots showing median metal concentrations and outliers for Sr, Ti, Ba, Pb, V and Cd (data separated by site and sites grouped by category; n = 30 for each site).*

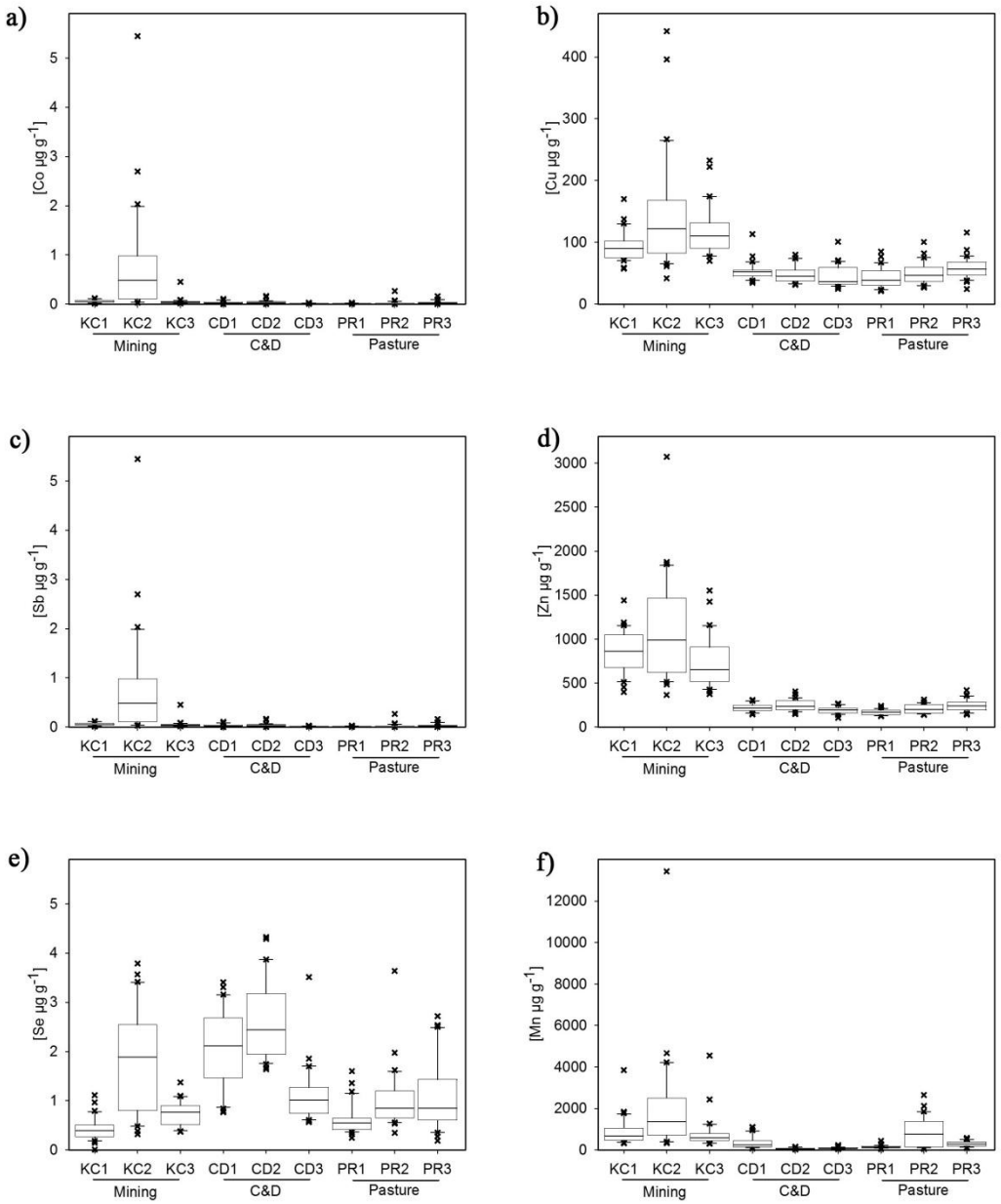

Fig. 4.4 *(a-f) Boxplots showing median metal concentrations and outliers for Co, Cu, Sb, Zn, Se and Mn (data separated by site and sites grouped by category; n = 30 for each site).*

The growth reducing effect of Ni, even at very low concentrations (Niyogi *et al*., 2014), may also exaggerate concentrations of other metals (Metcalfe-Smith *et al*., 1996; Lobel *et al*., 1991). While it is possible that the metals may have originated from painted (Ti) and preserved (Ni) timber fencing and metal gates (Weber *et al*., 2002; Kim and Chung, 2001) on the pasture sites (such livestock enclosures were about 20 m from the collection areas on these pastures, rather than over 30 m on mine and C&D waste sites), however this is unlikely, due to the limited (usually < 15 m in six months) dispersal ability of gastropods (Aubry *et al*., 2006; Baur, 1993). In addition, the accumulation of metals in plants (gastropod food source) is specific to both the metal and plant species (Deng *et al*., 2004) on each site and is influenced by soil parameters (Boshoff *et al*., 2013) such as pH, so it may be possible that conditions were more conducive (such as having a different pH or free metal ion concentration) to uptake of these particular metals on the pasture sites. The significantly elevated concentrations of Ag found in slugs from the pasture (compared to those from C&D waste), may be exaggerated by the slow excretion rates of Ag in gastropods (Croteau *et al*., 2011). It is worth noting that Ag (Fig 4.2) was found to be below the limit of detection for many samples on both C&D waste and unimproved pasture, and so this difference may be limited in it significance. In addition, Ag concentrations observed on pasture sites are similar to a previous study (control sites of varying habitats about 200m from contaminated locations) using the terrestrial snail *Cepaea nemoralis* (Linnaeus, 1758) (Boshoff *et al*., 2013).

The concentrations of Zn and Cu observed in this study (for mines) concur with previous studies that used *D. reticulatum* (Table 4.5) collected on contaminated sites (Greville and Morgan 1989b; 1990; 1991). Some studies investigated seasonal variation in metal concentration (Greville and Morgan 1989b), influence of slug size on concentrations (Greville and Morgan, 1990) and dermal absorption efficiency of metals (Bullock *et al*., 1992). Metal concentration were found to vary both seasonally (most likely as a result of activity levels) (Greville and Morgan, 1989b) and with size (Greville and Morgan, 1990), however the slugs used in the present study were all mature adults of similar size. Slugs used in laboratory studies (examining metal storage and toxicity effects (Triebskorn and

Köhler, 1996; Köhler and Triebskorn, 1998), accumulation strategies (Gräff *et al*., 1997) and stress reactions (Köhler *et al*., 1996)) exhibited more extreme concentrations for Zn (Triebskorn and Köhler, 1996) (higher in contaminated samples and lower in control samples), more than likely a result of the absence of other metals. The concentrations of Cd found during the present study are slightly lower than the concentrations recorded in other field studies from contaminated sites (Greville and Morgan, 1989a; 1989b; 1990; 1991). Sphalerite, a Zn containing mineral mined at Silvermines (Office of Environmental Enforcement, 2004) and Tynagh (Brogan, 2003), is associated with low Cd concentrations (Hem, 1989). as evident from Cd concentrations close to the limit of detection in groundwater from KC2 (even when other elements were present in high concentrations) (Henry, 2014). For one site in this study (KC2), Pb concentrations were found to have the same high concentrations as previous field studies (on mines) (Greville and Morgan 1989a; 1989b; 1990; 1991; 1993).

Although there is still some between-site variability, it is likely that slug metal content is a true reflection of the actual soil metal content across all sites. The variation may be partly attributed (especially on sites without obvious metal sources) to the known (Spurgeon *et al*., 2006) competition of free ions (other metals and H^+) for uptake pathways. In the case of KC2, which shows significantly ($P < 0.05$) elevated concentrations for Ba, Co, Mn, Pb, Sb, Tl and V relative to the other mine sites, these increased soil metal concentrations are likely associated with the onsite smelting plant (Spurgeon *et al*., 2006) as seen near smelting plants on other mines (Alloway, 2013).

Although molluscs are unable to regulate the uptake of metals via the digestive tract or through direct dermal contact (Gräff *et al*., 1997), they may cease or slow down food consumption, or avoid contaminated foods (Lefcort *et al*., 2004). In this situation however, dermal absorption may continue to increase the metal burden of the slug (Gräff *et al*., 1997). Concentration factors (ratio of metal concentration in slugs compared to the vegetation) have been shown to decrease as the metal concentrations increase, with the exception of Pb (Gräff *et al*., 1997). This suggests that care must be taken when attempting to determine precise metal

concentrations in soil and vegetation based on *D. reticulatum* metal concentrations. However, metal concentrations (Cd, Pb, Zn) in *D. reticulatum* increase as concentrations increase in their food source (Gräff *et al*., 1997). This would suggest that slug metal concentrations should reflect ambient metal concentration trends in the surrounding vegetation, and also that metal accumulation in these plants impacts on slug metal concentrations. Although sub-lethal concentrations of some metals (such as Cd and Zn) are known to cause sub-cellular damage (e.g. nucleolus alteration, mitochondrial swelling and microvilli shortening) in gastropods (Hödl *et al*., 2010), including *D. reticulatum* (Triebskorn and Köhler, 1996), detoxification is utilised as a survival strategy (Triebskorn and Köhler, 1996; Dallinger *et al*., 1989). This detoxification can involve either immobilisation (e.g. activation of metal-binding proteins such as metallothionein for Cd and Zn; Vijver *et al*., 2004) within cell lysosomes or precipitation into granules (Pb; Marigómez *et al*., 2002) which can be excreted via faeces (Triebskorn and Köhler, 1996; Dallinger *et al*., 1989; Desouky, 2006). Detoxification is metal specific, with essential metals, needed for biological functions, having the most efficient rates (Luoma and Rainbow, 2005). The non-essential Cd and Pb have no known function in biological systems and tend to be more poorly regulated (Vijver *et al*., 2004). Growth rates of molluscs are also known to be reduced by elevated concentrations of some metals (Cd, Cu, Ni; Niyogi *et al*., 2014; Dorgelo *et al*., 1995; Das and Khangarot, 2011), which in turn may increase overall metal concentrations, because the potential dilution effect of newly generated tissue is reduced (Lobel *et al*., 1991; Metcalfe-Smith *et al*., 1996). This may, therefore, compound slug metal content in contaminated sites. The accumulation of metals (Cd, Cu, Cr, Ni, Pb and Zn) in some bivalve molluscs is also known to be lowered by improving nutritional and physiological conditions (Voets *et al*., 2009; Martins *et al*., 2011; Mumbiana *et al*., 2006).

The uptake rate of metals by invertebrates is species specific, and dependant on a number of biotic (e.g. feeding behaviour and physiology; Peijnenburg, 2002) and abiotic factors (e.g. metal speciation, temperature and pH; Bullock *et al*., 1992; Fritoff *et al*., 2005; Peijnenburg, 2002; Spurgeon *et al*., 2006). In addition, the interactions of such biotic and abiotic factors with site specific characteristics

would influence metal uptake. The different types of ligand bonded to metals in food allow varying success in converting the metal to a bioavailable form (Peijnenburg, 2002). The speciation and solubility of metals, with free metal ions being the most significant in terms of bioavailability (Spurgeon *et al*., 2006; Hough *et al*., 2005), are strongly influenced by soil pH, although soil pH is not always correlated with invertebrate metal concentrations (Spurgeon *et al*., 2006). The availability of H^+, Ca^+ and Mg^+ ions is thought to reduce the uptake of free metal ions by blocking the membrane ligand uptake pathways (Spurgeon *et al*., 2006). The ratio of different metals is also important, where some metals may compete for cell membrane sorption sites, or have a synergistic effect if they share the same route (Dallinger *et al*., 2001; Hough *et al*., 2005).

Although the variable nature of C&D waste means that assumptions should not be made about which metals may be elevated at any single site, the present study highlights the bioaccumulation of priority metal pollutants (As, Ba, Cd, Co, Mo, Sb, Se, Sr and Tl) in slugs at C&D waste sites. While slugs from C&D waste sites should not have the same apparent degree of exposure to hazardous metals as slugs from mine sites, it is important to note that potential mobilisation of metals is not a criterion considered when waste licenses are issued for the disposal of such waste in the EU and elsewhere. Leachate from C&D waste is known (Roussat *et al*., 2008; Weber *et al*., 2002; Melendez 1996; López and Lobo, 2014) to contain elevated concentrations of metals (Al, As, Cd, Cu, Fe, Mn, Pb) and other contaminants (methylene chloride, 1,2-dichloroethane, sulphate, total dissolved solids). One of the most important and useful aspects of utilising biomonitors in environmental monitoring and assessment is that their tissues provide quantitative information on the bioavailable fraction of contaminants, which also have the potential for biomagnification in the food chain. Many of the difficulties associated with obtaining representative soil and groundwater samples from a heterogeneous matrix (such as C&D waste) are also avoided. This study has identified that certain metals known to attain high concentrations in C&D leachate, including As and Cd, are bioavailable, and therefore ecotoxicologically relevant. This can result in significantly elevated concentrations in gastropods on

Table 4.5 *Comparison of available published metal data (µg g^{-1} dry weight) in Deroceras reticulatum .*

Reference	Location	Pb	Cd	Zn	Cu	Fe (and Fe compounds)	Comment
Greville and Morgan, 1989a; 1989b	UK (Rhondda Cynon Taf)	130 ± 15	65 ± 10	900 ± 100	70 ± 15	-	Contaminated site (mine) (September data)
Greville and Morgan, 1990[35]	UK (Rhondda Cynon Taf)	130 ± 15.2	64.2 ± 10.4	874.9 ± 122.1	68.8 ± 16	-	Contaminated site (mine)
Greville and Morgan, 1991[25]	UK (Rhondda Cynon Taf)	162.4 ± 21.6	65.1 ± 17.5	735.2 ± 119.6	-	-	Contaminated site (mine)
	UK (Vale of Glamorgan)	3.2 ± 0.3	9.3 ± 0.8	205.2 ± 32.1	-	-	Control site
Bullock *et al*., 1992[32]	Laboratory	-	-	-	-	2554 ± 140.5	Exposed to Contamination
	UK (Hertfordshire)	-	-	-	-	103 ± 5.2	Irrigated field
Greville and Morgan, 1993[48]	UK (Rhondda Cynon Taf)	62.5 ± 10	-	-	-	-	Contaminated site (mine)
	UK (Vale of Glamorgan)	4.7 ± 1.9	-	-	-	-	Control site
Graff *et al*., 1996[43]	Laboratory	1168.6 ± 1532.3	245.9 ± 135	4252.4 ± 2785			Contaminated
	Laboratory	7.6 ± 9.3	7.8 ± 8.9	92.8 ± 69.1	-	-	Control
Köhler *et al*., 1996[71];	Laboratory	1168.6 ± 1532.3	245.9 ± 135	4252.4 ± 2785	-	-	Contaminated
Triebskorn and Köhler,	Laboratory	178.7 ± 115.2	121.8 ± 7.6	393.1 ± 192.0	-	-	Medium contamination
1996[33]; Köhler and Triebskorn, 1998[70]	Laboratory	4.4 ± 7.2	2.9 ± 1.4	76.1 ± 26.5	-	-	Control
This study	Ireland (Galway)	0.47 ± 0.35	4.86 ± 2.81	220.7 ± 47.2	53.3 ± 14.6	91.9 ± 25.8	C&D (CD1)
	Ireland (Galway)	0.65 ± 0.38	8.65 ± 2.88	246.4 ± 64.4	48.4 ± 13.8	82.2 ± 20.3	C&D (CD2)
	Ireland (Galway)	0.22 ± 0.17	8.63 ± 5.35	193.6 ± 39.3	43.9 ± 17.4	100.2 ± 69.6	C&D (CD3)
	Ireland (Galway)	0.42 ± 0.14	2.35 ± 1.22	170.0 ± 31.9	43.2 ± 16.1	110.8 ± 30.5	Unimproved pasture (PR1)
	Ireland (Galway)	0.41 ± 0.32	5.04 ± 3.05	206.8 ± 50.1	49.6 ± 17.2	90.5 ± 16.9	Unimproved pasture (PR2)
	Ireland (Galway)	0.30 ± 0.14	4.89 ± 2.26	246.2 ± 66.7	58.7 ± 17.8	111.5 ± 36.0	Unimproved pasture (PR3)
	Ireland (Tipperary)	5.74 ± 3.62	42.9 ± 19.4	857.5 ± 249.3	92.9 ± 24.2	80.8 ± 34.5	Mine (KC1)
	Ireland (Tipperary)	274.2 ± 215.2	34.4 ± 18.5	1086.1 ± 573.2	144.7 ± 91.6	298.4 ± 338	Mine (KC2)
	Ireland (Galway)	8.29 ± 6.27	51.0 ± 45.7	742.6 ± 301.1	120.0 ± 40.0	90.8 ± 43.2	Mine (KC3)

Concentrations expressed as mean ± standard deviation

C&D waste, compared to the baseline unimproved pasture.

The potential risks of metal contamination in the biota at higher trophic levels or adjacent to such waste are not yet known, although terrestrial biomagnification is known to occur (Van Straalen and Ernst, 1991). Insectivores such as *Erinaceus europaeus* L. (Rautio *et al*., 2010), are known to be sensitive to diet-borne metal accumulation. Any potential contamination threat to adjacent areas would likely be site-dependent, with variables such as waste contents (Weber *et al*., 2002; Melendez, 1996), geology (Homoncik *et al*., 2010; Aziz *et al*., 2008), soil (Peijnenburg, 2002), recharge, aspect (direction of surface runoff) and groundwater flow likely to be influential factors. Limestone in particular is known to buffer and aid precipitation of metals (Aziz *et al*., 2008) from leachate and waters, and so is likely to reduce risk (compared to other non-carbonate bedrock) to drinking water supplies which are not immediately adjacent to the waste. The elevated metal concentrations in the leachate (as evident from previous studies) and slugs from C&D waste indicate that action should be taken to minimise the risk of future contamination. By separating waste more efficiently (such as source/on-site separation; Tam and Tam, 2008; Poon *et al*., 2004; Teo and Loosemore, 2001) and diverting more C&D waste to recycling (Duran *et al*., 2006; Coelho and DeBrito, 2013), the dependency on disposal would be reduced, meaning fewer infill sites would be required (Symonds Group Ltd. *et al*., 1999). This improved separation would also be likely to reduce the amount of unpermitted items/substances that occur in C&D waste infill.

4.5. Conclusions

This study demonstrates the potential usefulness of employing *D. reticulatum* as a biomonitor of metals on C&D waste sites and has, for the first time, shown that gastropods collected on C&D waste have significantly higher metal concentrations than those from the unimproved pasture (for As, Ba, Cd, Co, Sb, Se and Tl), and mines (for Mo, Se and Sr). The most likely source of these EU priority pollutants is the C&D waste itself, although the exact sources of contamination within the waste are difficult to isolate due to the varied nature of the materials within each site (and even between regions or countries). Improved waste separation and recycling rates would be likely to reduce the number of infill sites and the volume of unpermitted items mixed through the waste. Unlike soil or

water analyses, these biomonitors reflect only the bioavailable (and so ecotoxicologically important) forms of metal, indicating the importance of such monitoring. This study is a first, from which other studies around the world can be compared. It has highlighted the need for further investigation into the bioaccumulation (and potential biomagnification) of metals in the biota from such C&D waste infill sites, which are common and often unregulated throughout the world. Where metals are found to be bioaccumulating in organisms to dangerous concentrations, there is a need for better enforcement of existing environmental protection policies and possibly implementing policy changes to reduce the impact of such sites on other environmental compartments and to encourage more sustainable development in the future.

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Chapter 5:

General Discussion

5.1. General Discussion

Although C&D waste is produced in very large quantities around the world (Schrör, 2011; USEPA, 2009), relatively little research on the environmental impact of using the waste as an infill material for wetlands has been completed, largely as the waste is perceived as being generally inert. This project is the first work of its kind to assess the nature and extent to which C&D waste infilling is carried out at a local scale, in addition to investigating the ecological and bio-contaminative implications of infilling wetlands with the waste. Local scale studies can provide useful data, the detail of which is not currently available in the international literature. In addition, it is hoped that the results of this study for a single local authority will encourage other local authorities to examine their C&D waste disposal practices and incorporate the recommendations from this study in their county development plans.

5.1.1. Key Findings

The main findings from the previous three chapters of this thesis are discussed below in relation to the overall study aims, which were to:

1. Assess the spatio-temporal distribution patterns of C&D waste infill sites, both legal and illegal, and identify issues with non-compliance on legal sites.

2. Assess the qualitative and quantitative impacts of infilling wetlands with C&D waste on soil, and plant and dipteran communities.

3. Examine the use of the Grey Field Slug (*Deroceras reticulatum*) as a potential biomonitor on C&D waste of metals, including priority pollutants with known biotic toxicity.

Recommendations are made from the findings of this study to inform best practice in the management of C&D waste disposal and existing site monitoring. The limitations of the study and suggestions for future research are also presented.

There are six key findings from this study:

i) The distribution of C&D waste infill sites in the local authority case study is focused primarily around large urban areas, major road networks and SACs, with wet grassland and peatland the most common habitats infilled.

The spatio-temporal distribution patterns of C&D waste infill sites has, to date, not been examined. A study in the United States during the 1990s found that there were approximately 1,900 documented C&D waste landfill sites, though information on site location and habitats lost were not given (Franklin Associates, 1998). For the present study, sites were found to be concentrated around large urban centres and major road networks. This was not unforeseen, due to the main sources of C&D waste being urban areas, and main roads allowing easiest transport of the waste with large machinery. This does, however, mean that wetlands located near urban areas and large roads are potentially at an increased risk of damage or loss due to infilling. For this reason, local authorities should be vigilant to ensure that wetlands in these areas, particularly those that are small and discrete, are surveyed prior to infilling to assess their ecological importance, even if they do not have a European or national designation of importance.

The location of over 40% of sites (of n = 84) within 1 km of SACs is a major cause for concern. In addition, four sites were found to be on or immediately adjacent to SACs, with a further four adjacent to NHAs. It is particularly important that such protected wetlands are not threatened by C&D waste infilling activities, either directly (through infilling) or indirectly through contamination and impacts on hydrological regimes. To ensure this, a full Appropriate Assessment (AA) should always be carried out where the wetland is within 15kms of an SAC (DEHLG, 2009) in conjunction with a detailed ecological survey (flora, fauna and hydrology). Sites located within this area that currently have a waste permit (without initially carrying out an AA) should also have a full AA undertaken before any permit renewals are granted. Ecosystem services such as habitat provision, flood control and water filtration (Mitch and Gosselink, 2007; Haines-Young and Potschin, 2010) associated with such valuable (environmentally and economically) wetlands are known to be more valuable than the post-infilling, improved land (Balmford et al., 2002). Costanza et al. (1997) valued wetlands (using global averages) to be worth almost US$ 15,000ha^{-1} based

on their ecosystem services including waste treatment and water regulation, compared to just under US$ 1,000ha^{-1} for woodlands. Balmford et al. (2002) reported that the value of Canadian wetlands (including their ecosystem services) were reduced from over US$ 16,000ha^{-1} to less than US$ 8,000ha^{-1} when they were used for intensive agriculture. That this study found these habitats (and associated ecosystem services) to be significantly modified post-infilling adds to the concern for the protection of SACs and any other important wetlands nearby.

The majority (57 %) of permitted C&D waste infill sites studied (n = 84) occurred on wet grassland habitat, followed by peatlands (23 %), quarries (2 %) and dry grassland (1 %). The remainder were on various habitat mosaics. The eastern half of County Galway has the highest population density (Central Statistics Office, 2011) and although there are areas of exception, in general it contains more grassland habitats than the western half, which is predominantly peatland (EPA, 2000). Knowing that the distribution of infill sites tends to be focused around the urban centres with high populations, it is therefore not unexpected for more wet grasslands to be infilled than peatlands in County Galway. Future losses of such habitats should be minimised as much as possible through waste reduction on-site (education of construction industry workforce and encouraging low waste building methods such as off site prefabrication (Lu and Yuan, 2013)) and increased recycling rates (taxing virgin raw materials and subsidising recycled aggregates), particularly as wetland coverage in the case study area is known to have declined recently (EPA, 2000). Fig 5.1 briefly summarises the preferred methods to tackle C&D waste and preferred site selection for disposal.

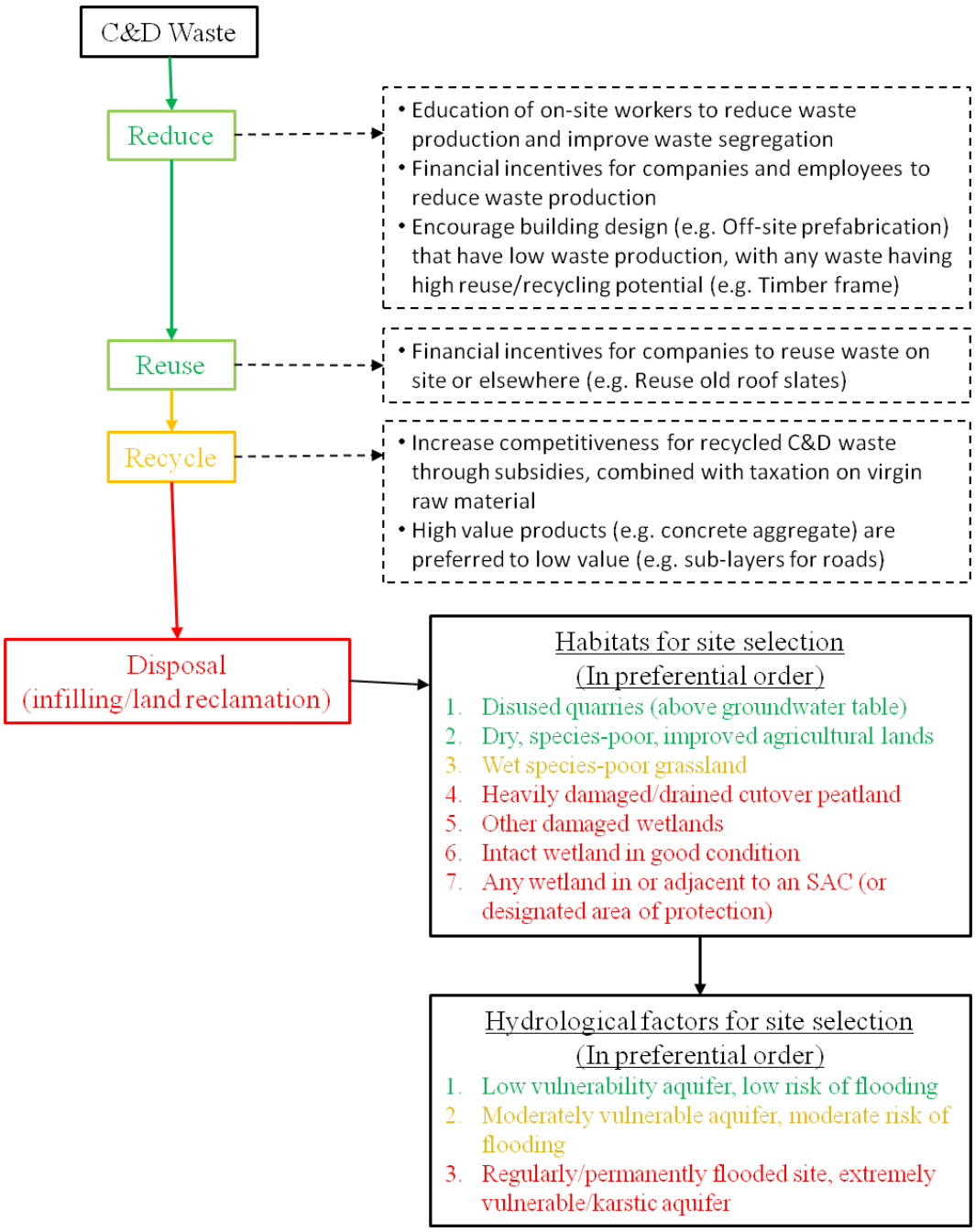

***Fig 5.1**. Brief overview of preferred treatment methods for C&D waste, and preferred site selection for its disposal. Preference shown by colour: Green = most preferred, Amber = less preferred, Red = least preferred.*

ii) Undocumented infill sites appear to be more prolific than sites with permits near urban areas, and non-compliance with permit conditions is widespread.

Unregulated C&D waste infill sites are common in the United States (Franklin Associates, 1998; USEPA, 2014), with the state of Georgia alone having an estimated 900 illegal sites in the mid-1990s (ICF Incorporated, 1995). At the same time, the total number of permitted C&D waste specific disposal sites across the whole United States was just 1900 (Franklin Associates, 1998). Many European countries such as Hungary, Spain and Italy also have some known problems with undocumented infilling, however the abundance of such sites there has not been published (BIO Intelligence Service, 2011; Symonds Group, 1999). This current study has provided an estimate of the density (number of sites per km^2) of undocumented sites close to main urban areas in County Galway, Ireland as a case study.

It was found that the density of these undocumented sites is greater than the documented sites within 5 km of urban areas. This is cause for concern on several levels. It shows that wetland loss due to infilling may be more widespread than anticipated, and also it suggests that there is a disregard for the requirement to have a permit and/or ignorance of the need to have a permit. This should encourage policy makers and governing bodies (such as the EPA) throughout Europe to undertake larger surveys on undocumented infilling activities, and to equip local authorities with the resources to ensure landowners are aware of the need for a permit for infilling activities and to ensure that such illegal activities are prevented.

For the sites with permits, non-compliance with permit conditions was found to be a widespread problem, with infilling limits (volume and area) not being adhered to (on 35% of sites which were granted permits), waste being contaminated with unpermitted materials (on 43% of sites visited during this study), and infilling being carried out prior to receiving (or after expiration of) valid permits. Although site monitoring and the use of warning letters has helped to reach a resolution on a small number of sites, these issues of non-compliance are likely to be heightened by both the apparent lack of consequences for such actions and the perceived

inertness of the waste. A suggested way to tackle these problems is ensuring local authorities have the resources to tackle the problem with a combination of educating permit holders and policing sites effectively.

iii) Soil properties and plant communities of wetlands post-infilling with C&D waste are clearly different to adjacent intact wetlands.

The soil properties of sites post infilling were radically different from the adjacent wetlands, something which is visibly noticeable on sites. The mainly organic soils of the wetlands, which had high moisture contents, were replaced with mainly inorganic, low moisture content soils, with the pH also increasing (from 6.41 to 7.94). Plant communities changed from primarily wetland species to ruderal species, mainly due to the different soil parameters, although disturbance caused by infilling activity would have aided the latter's rapid colonisation of the sites. Although species richness was higher on the C&D waste infill, the commonness of these ruderal species means that they would be much less favoured, from an ecological perspective, than wetland specialists, which provide many wetland ecosystem services. The dominant presence of *Agrostis stolonifera*, *Circium arvense*, *Holcus lanatus* and *Ranunculus repens* on many infill sites is supported by their preferred habitat of disturbed ground (Grime et al., 1996). Similarly, the dominance of species such as *Molinea caerulea*, *Potentilla erecta* and *Cladium mariscus* in different wetland habitats conforms with their known preferences for high moisture soils (Grime et al., 1996). The dramatic change in soil parameters and plant communities shows that the ecological conditions on sites are altered post-infilling, with the loss of many of the wetland specific species and their associated ecosystem services, which include water filtration and nutrient removal (Mitsch and Gosselink, 2007; Haines-Young and Potschin, 2010), particularly for plants such as *Iris pseudacorus* and *Phragmites australis* (Wu et al., 2010). This efficient nutrient removal by wetland plant species is also the basis of constructed wetlands which are used to effectively treat wastewater (Kadlec and Wallace, 2008). In place of these wetland plant species are more ruderal species, which from a conservation perspective are less advantageous due to their frequency and abundance around Ireland. It is likely that the addition of any amount of C&D

waste to an area of wetland will dramatically change the biota of that area, so it is important that the most intact wetlands are preserved as a priority. Carrying out a detailed botanical survey (together with zoological and hydrological studies) before any permits are issued would allow the identification of such intact wetlands, and any particularly important habitats. Where infilling does have to occur, a mitigation strategy can then be drawn for the site to preserve as much of that habitat as possible.

iv) Aerial invertebrate communities from C&D waste infill are significantly different from the adjacent wetlands.

Changes to dipteran communities as a result of habitat modification, particularly to vegetation structure, have been documented (Hughes et al., 2000; King & Brazner, 1999; Whiles and Goldowitz, 2001). Sciomyzids in particular are known to show a high sensitivity to wetland condition (Murphy et al., 2012; Speight, 1986; Williams et al., 2009, 2010). The dipteran families represented in this study were found to have changed post-infilling with wetland specialists, such as Culicidae and Chironomidae, being reduced significantly. This is most likely as a result of the different plant communities present and the associated different physical structure on the infill, in addition to the decreased soil moisture and reduced permanent surface water areas. Such a strong reduction in sciomyzid abundance and species richness on the C&D waste was unanticipated, and their usefulness as bioindicators (Williams et al., 2009; 2010) of habitat change is strongly supported by this study. In addition, such clear difference between sciomyzid communities of these different habitats in close proximity (< 15 m) reinforce the theory that sciomyzid ranges are restricted to small areas. These findings should also encourage the implementation of thorough environmental assessments (which include invertebrates) prior to infilling. Such data would not only allow for useful comparisons to the post-infilling plant and invertebrate communities, so expanding this study, but would also allow for identification of any rare species present on-site so mitigation measures could be put in place.

v) The slug *Deroceras reticulatum* collected on C&D waste infill contains significantly higher levels of certain priority pollutant metals (known to be potentially toxic and hazardous) than those collected on control sites.

Individuals of *Deroceras reticulatum* collected on wetlands that have been infilled with C&D waste were shown to have higher tissue concentrations of priority pollutant (Council Directive 2008/105/EC; Council Directive 2006/11/EC) metals As, Ba, Cd, Co, Sb, Se and Tl than those collected from control sites (extensively farmed pasture). This indicates that such toxic and hazardous metals contained in C&D waste can accumulate in a biomonitor species, which could be utilised for future studies and site assessments as an alternative to direct soil and water analysis. The use of biomonitors removes any logistical difficulties of soil and water sampling and provides information on bioaccumulation and potential biomagnification. Previous studies had shown that the leachate of C&D waste contained elevated concentrations of Al, As, Cd, Cu, Fe, Mn and Pb (Weber et al., 2002; Roussat et al., 2008; Melendez, 1996; Lopez and Lobo, 2014), but it was not known if these metals were in a bioavailable form affecting invertebrates. It was also shown in previous studies that *D. reticulatum* tissue can accumulate metals including Pb, Cd, Zn and Cu from its environment (Greville and Morgan, 1989; 1990; 1991; Köhler and Treibskorn, 1998). This potential metal contamination is of particular concern as the spatial distribution of sites was found to be significantly concentrated around SACs (which may be potentially affected by water contamination, bioaccumulation and biomagnification), and 54% of sites were found on extremely vulnerable aquifers. Although the potential impacts of C&D waste on such aquifers are not yet understood, the presence of elevated metal concentrations in its leachate (Lopez and Lobo, 2014; Melendez, 1996; Roussat et al., 2008; Weber et al., 2002) and biota means that caution should be exercised on site location. The metal within the gastropod tissue may have the potential to be passed along the food chain and even undergo biomagnification (van Straalen and Ernst, 1991) in various predators such as general insectivores (e.g. *Erinaceus europaeus* L.; Rautio *et al*., 2010) and species specific predators and parasitoids such as the sciomyzid, *Tetanocera elata* (Fab.) (Barker *et al*., 2004; Knutson and Vala, 2011).

vi) There are many challenges facing scientific investigations of C&D waste infill sites

Due to the environmentally sensitive nature of infilling wetlands with C&D waste, any studies being carried out on such sites are likely to face various challenges. Initially, obtaining access permission from landowners may be difficult. It was found that assuring landowners that site specific data remained anonymous was the most effective approach. One of the main problems encountered throughout this study was disturbance of equipment on-site by large site machinery and members of the public. Their location near urban areas and main roads means that any equipment is likely to be highly visible to the public. Ensuring that any equipment left on site is not immediately apparent should help to reduce this problem (through the use of inconspicuous colours, with low profile structures such as miniature malaise or SLAM traps), along with increasing sample collection frequency to prevent long term sample loss. However, the most ideal solution would be choosing a sampling strategy that minimises the need to leave equipment on-site (such as using a sweep net or suction sampler for invertebrates where possible). Equipment choices and methodologies for any sampling activities are also often greatly restricted as a result of the attributes of the sites and the waste therein. Placing equipment such as posts or piezometers into the waste can be difficult due to the nature of the C&D waste substrate which can be a compact, poorly sorted heterogeneous mix (ranging in size from clay to cobbles and boulders). Surface topography was often hazardous for walking or sampling, and the waste often had protruding items such as steel bars. All of these factors should be accounted for in future studies during the planning phase to save time and effort, and to ensure the most effective sampling methodologies are used. In such unfavourable site conditions, the use of pan traps, miniature malaise traps and suction samplers could be considered with frequent or short, staggered collections (so traps are not always on site). These problems may be reduced for future site monitoring with increased enforcement of the correct site security measures (particularly for equipment interfeerence) and surface finishing (levelling and planting) post-infilling.

5.2. Recommendations

There are a number of recommendations to be made following this study:

- Local authorities should be given the resources to implement an education programme for all permit applicants, to ensure they are cognisant of the

environmental and legal consequences to breaking the conditions of their permit. The general public should also be made aware of the environmental consequences to infilling wetlands with C&D waste.

- Local authorities need sufficient resources to effectively monitor and police infill sites to ensure permit terms are adhered to.
- All applications made for infilling permits should require detailed ecological surveys (including botanical, invertebrates, vertebrates and hydrological) regardless of the site size or location and, where the site is within 15km of an SAC, a full AA should be undertaken. This will ensure that valuable wetland habitats are not lost. Permit terms should also include a stipulation that allows future environmental studies to be carried out on the sites, and if required (for assessing hydrological parameters), the installation of hydrological piezometers before completion of the infilling process. This initial survey (and AA) should also serve as a baseline for such studies, to effectively assess any impact that the C&D waste has had on the wetland. Lining C&D waste infill sites and treating the collected leachate would be the most effective method for minimising the contamination risk of the waste to groundwater.
- Mitigation strategies should be built into each permit, ensuring that the most ecologically valuable and hydrologically sensitive areas of intact wetland, which may be locally important, remain as they are. This will ensure that their associated biodiversity and ecosystem services are not completely lost. For these sensitive areas, there is no 'safe' amount of C&D waste that can be used, as the ecological communities will likely change everywhere that is infilled.
- In addition to EU and national waste policies, Local Authorities should have a clear strategy for C&D waste disposal. This should aim to reduce production and increase recycling rates through: educating the construction sector workforce; introducing financial incentives for low waste building methods such as prefabricated housing; taxation of raw materials and subsidies for recycled materials. In addition it should also ensure that the most environmentally sensitive areas in that local authorities' area are protected from infilling activities.

- Future studies on C&D waste should ensure their research strategies allow for the challenges they are likely to face from the outset. This could be done by choosing field methodologies (as mentioned in section 5.1) that are suited to site conditions, and minimising the need for equipment to be left on site permanently.

5.3. Limitations of this study

This study used the Local Authority area of County Galway as a representative case study to assess infilling wetlands with C&D waste. There are, however, some limitations to this study to consider:

- Pan traps were the only invertebrate sampling method to be used across all study sites in Chapter 3, while sweep nets were only used on one site. Certain invertebrate surveying techniques, such as emergence and malaise traps (full size), were not practical as they would attract undue attention from members of the public. In addition, large numbers of any smaller traps may have the same effect. Site topography was in some cases quite hazardous, and would not allow for certain invertebrate capture methods, such as sweep netting. This did place some restrictions on the analysis, but the obtained dataset, which shows many clear differences between infill and wetland areas, is nonetheless likely to be a good representation of the on-site biota. Carrying out sampling transects along the wetland-infill gradient would be useful, but this was not practical on the sites in this study for safety (water depth increased rapidly in some wetlands) and practical (site size) reasons.
- For 2009, the beginning of the summer season was not ‘captured’ using the invertebrate sampling techniques. This was due to the inability to obtain permission to access sites until midway through the summer. It did not, however, impact on the aims, and the complete season was ‘captured’ in 2010.
- The monitoring of hydrological parameters could not be carried out due to the associated cost involved with detailed analysis, and the difficulty involved with installing piezometers in C&D waste post-infilling. This would have added an additional element to the biomonitor metal analysis. In addition, any

impacts on the hydrological regime of a wetland are likely to affect the ecological communities therein.

5.4. Suggested future research

- The density and distribution patterns of both documented and undocumented infill sites should be assessed throughout Ireland and internationally. This would allow identification of the regions and wetlands that are most under pressure from infilling activities.
- In order to carry out a more representative international study, similar studies of C&D waste infill sites should be carried out in representative areas of other countries around the world. This can be applied to both ecological and contaminative impact perspectives.
- Assessing the impact of C&D waste infilling on additional invertebrate species and communities (such as molluscs, Carabidae and Formicidae) to obtain a more comprehensive knowledge of how the activity is effecting the biodiversity of such wetlands. The use of pitfall traps for groups such as Carabidae would be limited on sites that are prone to regular surface flooding, and would require difficult digging on the C&D waste. The use of transects would allow assessment of what distance C&D waste may affect ecological communities.
- Assessing metal accumulation rates from C&D waste in plants (such as grasses and herbaceous annuals which are eaten by slugs) and additional invertebrate species such as the earthworm (*Lumbricus terrestris*) would greatly help in discovering the potential contaminative effects of the waste in wetlands. This knowledge could then be used to create policy and legislation to encourage increased recycling and ensure that waste is treated in a more appropriate manner if required.
- Carrying out detailed hydrological surveys and analysis of infill sites using multiple simple piezometers, prior to the infill activity and post-infilling. This will provide information relating to the impact of the waste on (ground and surface) water chemistry and wetland hydrological regimes, and any potential impact that infilling may have on adjacent wetland sites. This is particularly

important as so many sites were found to be adjacent to SACs. It would also be important to research the most effective methods for reducing the production rates of leachate from C&D waste (e.g. capping, lining, etc.).

5.5. References

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