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A natural capital approach to integrating science, economics and policy within decision making: Public and private sector payments for ecosystem services

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Introduction

Overview

We demonstrate how different payment mechanisms can stimulate the efficient delivery of key, high-value ecosystem services which are either not produced, or are under-produced, by the normal operation of the market. Two payment mechanisms are considered: payments from the public sector to private businesses; and payments between private businesses. Public to private funding provides the most common Payments for Ecosystem Services (PES) mechanism in the UK and most other countries. By contrast private to private (i.e. business to business) PES mechanisms remain relatively novel yet, because they tap into private sector funds, they have great potential for incentivising environmental improvements, particularly in cases where there is a profit opportunity arising from such improvements.

Permutations of these mechanisms are illustrated through three case studies: Public to private funding of natural capital improvements for national level decision making (referred to as the “national level case study”); Public to private funding of natural capital improvements at catchment level (the “catchment level case study”) and; Business to business funding of natural capital improvements again at a catchment level (the ‘business to business’ case study). Together these form a matrix of decision level and funding source exemplars which provide should have wide applicability.

Context: The natural capital approach

The UK and Natural Capital

The term natural capital refers to those stocks of assets, provided for free by nature which, either directly or indirectly, deliver wellbeing for humans. These assets include stocks of freshwater water, fertile soils, clean air and living things. Natural capital stocks in turn deliver flows of services, often called ecosystem services, which (often in combination with flows from other capital including human labour, ingenuity and manufactured goods) produce the benefits upon which humans depend for economic wellbeing and indeed their very existence.

Natural capital stocks include both renewable resources (such as fish populations which, with careful management can self-replenish permanently) and non-renewable assets (such as oil stocks, the proceeds of use needing to be in part reinvested if we wish to retain the services of such stocks, typically via

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investment in alternative sources of those services). Both stock types are vital contributors to economic activity and wellbeing and both assets can be driven to exhaustion through human action.

Economic activity therefore both draws and depends upon natural capital whilst also affecting the stock of those assets. This intimate relationship between the environment, the economy and human wellbeing has prompted the UK Government to adopt the overarching policy goal to be "the first generation to leave the natural environment in a better state than it inherited." (H.M. Government, 2011; see also House of Commons, 2012). As part of this ambition the UK has invested in research seeking to develop a 'natural capital approach' to decision making (*ibid.*; UK NEA, 2011; UK NEAFO, 2014), which explicitly recognises the dependence of economic value and wellbeing on the natural capital stocks provided by the environment and the ecosystem service flows which those assets provide. To help guide this process, the 2011 Natural Environment White Paper (H.M. Government, 2011) set up the world's first independent Natural Capital Committee (DEFRA, 2012; NCC, 2013) to advise on the restoration and improvement of natural capital as a means of sustaining and enhancing economic growth in the UK. Importantly, while it has a close relationship with the UK's environmental department (the Department for Environment Food and Rural Affairs; DEFRA, 2017a), the NCC actually reports to HM Treasury, the UK's finance ministry. Indeed, the UK's Chief Finance Minister, the Chancellor of the Exchequer, chairs the Economic Affairs Committee (EAC, 2017) which the NCC formally advises (NCC, 2017b). At an institutional level therefore the UK explicitly recognises the role of natural capital within economic growth. The NCC has reported extensively on methods to 'mainstream' consideration of natural capital into both policy and business decision making (NCC, 2017a,b). Furthermore, it has also provided extensive advice on the valuation, accounting and financing of natural capital enhancement (NCC, 2017b,c).

Mainstreaming Natural Capital: The Drivers of Change

Mainstreaming natural capital involves bringing nature's stock and flows of goods and services into decision making. A key element of this is to provide decision makers with an understanding of the factors that drive change in natural capital resource use. A common failing in the literature is that analyses consider the advantage of moving from current to alternative resource use without consideration of how the move between these two states is to be effected. For example, in the case of land-use it is relatively easy to demonstrate that a move from current intensive agricultural production practices to lower input systems will deliver improvements to water quality, greenhouse gas (GHG) emissions, wildlife habitat, greenspace access etc. These advantages are often rigorously demonstrated, however without guidance as to how such change should be delivered, the decision maker faces uncertainty regarding how best to act; indeed, arguably natural capital analyses are of little practical use without information about how a decision maker should affect change. Given this, the chapter begins with a case study explicitly tackling this challenge. It acknowledges that land-use change is driven by a wide array of forces which can be gathered together under three broad headings; socio-economic and market forces, policy drivers, and environmental drivers, as overviewed in Box 1.

Box 1: The Drivers of Change

Changes in human use of natural resources are driven by a variety of factors. These can be grouped as follows:

1. Socio-economic and market forces including:
    a. Demand and supply of goods
    b. Related prices of outputs and costs of inputs
    c. Technology and its impact on the production process, in particular its effects upon costs

2. Policy drivers including:

a. Regulations – e.g. The EU Drinking Water Directive which regulates the levels of pollutants deemed safe in drinking water.
b. Subsidies – e.g. The Common Agricultural Policy whose payments are in excess of £2.5billion per annum in the UK, a sum which represents more than half of farm income in a typical year (National Statistics, 2017).
c. Taxes – e.g. The Landfill Tax which is paid by a business to get rid of waste using landfill sites. The standard rate is currently £84.40 per tonne.
d. Payments for Ecosystem Services (PES) – e.g. Countryside Stewardship provides grants to foresters and land managers for creating and managing woodland to sustain and increase the public benefits from woodlands.
e. Other incentives – e.g. The European Union's Emissions Trading Scheme created a market to enable the trading of Greenhouse Gas emission allowances to reduce emissions across Europe.

3. Natural environment drivers including:
    a. Spatial variation (the changes in the natural environment and its ecosystem services across locations) including:
        i. Immediate environment of production area
        ii. Soil types
        iii. Rainfall
        iv. Temperature
        v. Wider environment (catchment conditions)

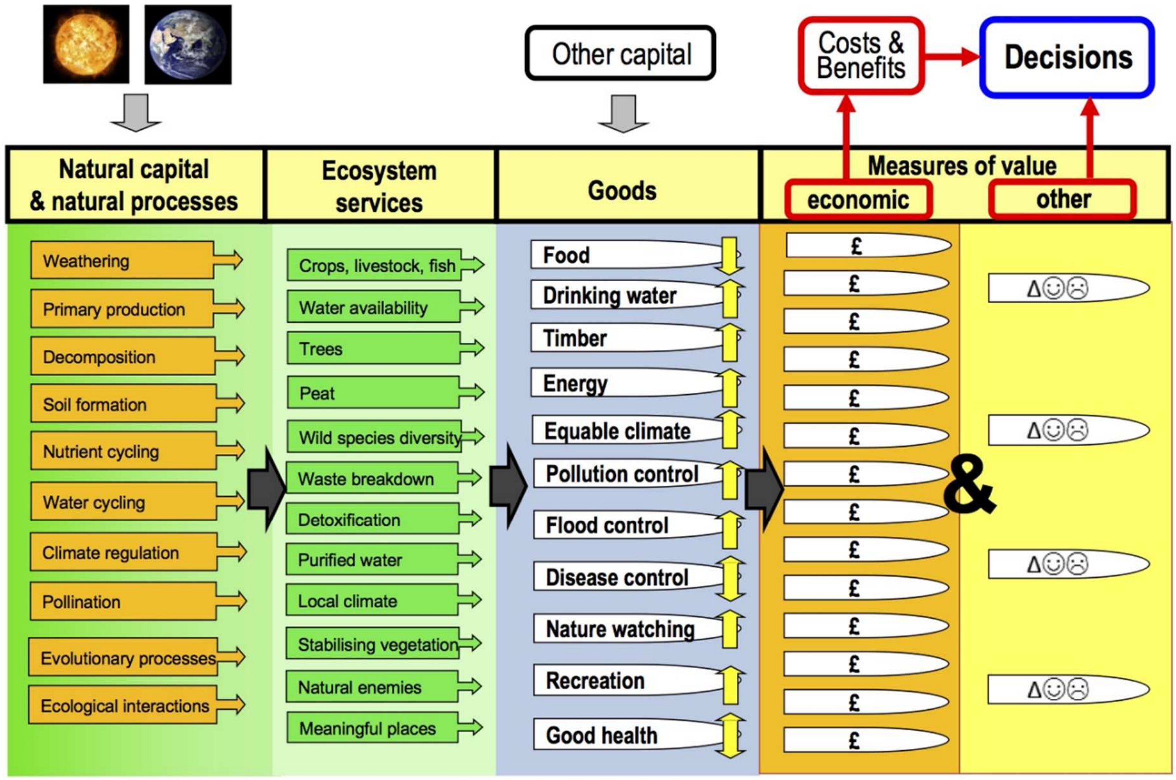
    b. Temporal variation (the changes in the natural environment and its ecosystem services across time) including:
        i. The dynamics of certain wild species populations such as pollinators.
        ii. On-going climate change and its myriad impacts

Any study that fails to provide analysis of the drivers of change simultaneously fails to provide decision makers with clear guidance regarding the strategies and policies they should adopt to bring about desirable change. Policy makers have to be able to understand how to effect a move from a current to a preferred state, otherwise knowledge of the advantages of such a change risks being impotent. Many of the studies given in the current literature focus on demonstrating desirable future environmental relationships, but fail to discuss the policy or other measures that should be adopted to achieve such outcomes. Understanding the drivers of change and the consequences brought about by policy decisions is one of the major reasons for bringing economists into decision making and it is arguably just as important as demonstrating the benefits of preferred states.

Natural Capital, Ecosystem Services, Goods and Values

When making policy decisions regarding the natural environment it is important to understand the linkage between the various forms of natural capital, the ecosystem services they provide and their transformation into valued goods and services. Figure 1 illustrates these connections. In the upper left of this Figure we have the raw inputs to this system: energy (e.g. from the sun) and matter (from the earth). Together these yield stocks of physical natural capital (e.g. soil) and natural processes (e.g. water cycling) (as shown in the first column of the figure). Combining these stocks and processes provides the myriad ecosystem service flows provided by the natural environment (second column). These services, such as the growth of trees or supplies of clean water, have the potential to directly provide certain wellbeing-bearing goods to humans. However (as shown in the third column), goods are more typically obtained by combining ecosystem service flows with other, human derived, forms of capital, such as labour, machinery and technology. Here the term 'goods' refers to anything which alters human wellbeing, ranging from tangible products such as timber or food, to non-tangibles such as the knowledge that other species are being sustainably conserved. Similarly while some of these goods are provided through markets and consequently have prices, others are provided outside markets and lack prices. Nonetheless all are, by definition, of value.

Figure 1: Decision Making and the Environment: From Natural Capital to Decisions.



Note: The yellow arrows illustrate the multiple effects typical of a change in natural capital, in this case those arising from an investment to establish woodland on a currently farmed area.

As discussed at greater length subsequently, because natural capital and ecosystem services can be used to generate a wide variety of goods, it is useful to understand whether those resources could be used in better ways. In effect we need some measure of how valuable a set of goods. Economic research over the past three or four decades means that many of the goods that contribute to human wellbeing can be assessed in economic values (fourth column), not just those with market prices, and changes in these can be analysed in terms of the resultant benefits and costs. However, a few wellbeing-baring goods cannot be robustly assessed in terms of economic value and therefore other, ideally quantitative, measures also have to be incorporated into decisions.

Figure 1 also illustrates the systems nature of changes to the natural environment. In its raw, unused state, natural capital resources are low entropy i.e. they have high usefulness and can be employed to generate a wide range of goods, often simultaneously. However, this means that changes to the use of natural capital often generate multiple consequences. The environment is an interconnected system, changing its use in one way can have multiple effects, many of which might have been unanticipated by the decision maker who prompted that original change. To illustrate this the yellow arrows in the central column of the Figure indicate the gains and losses resulting from a proposed investment to restore forests on farmland. Such afforestation of farmland will typically reduce the amount of food produced. If the analysis is curtailed at that extent then the investment might often appear to yield poor value; timber values are long delayed and may well be less than the food value that can be generated over that period. Such restricted analysis is common, especially if food and timber are the only marketed, and hence priced, goods produced by this change. However, afforestation can affect the production of a wide range of goods other than just increases in timber and decreases in food output. A shift from agriculture to woodland can often result in an improvement in water quality as forests require much less or even no inputs of fertiliser inputs than farmland. Conversion to woodland can significantly reduce the runoff of nutrients into waterways resulting in less polluted rivers and higher water quality. In very many cases woodlands also reduce emissions of air pollution and unlike intensive farming often store carbon helping reduce climate change. Similarly woodlands typically provide much greater recreational benefits than many forms of agriculture. These various effects, illustrated by the multiple yellow arrows in Figure 1, show that restricting analysis of the impacts of changes in natural capital usage to just those that have market prices, can lead to very misleading impressions of the value of change. To improve decisions regarding natural capital we need to assess all the major trade-offs arising from a proposed change and ensure that they are valued on a level playing field.

Decisions, trade-offs and Valuation

Two inescapable facts

The central challenge facing all decision making can be encapsulated within two inescapable facts:

1. Human wants (including those with the highest possible motivations such as improving society) exceed the resources available to satisfy them all;
2. Because of these resource constraints, every time we decide to do one thing, we are in effect making a decision not to do another; our decisions implicitly place values on each option.

These facts mean that trade-offs are inevitable which in turn means that valuations are unavoidable - they are the essence of decision making. The only real question is whether we leave those trade-offs and valuations implicit and often hidden within a decision, or instead make them explicit and open to scrutiny.

Making trade-offs and valuations explicit of course opens them up to scrutiny and challenge; forming a lightning rod for criticism. Economics analyses of environment related investments are frequently the focus of criticism precisely because they make their valuations clear. However, failing to reveal valuations does not mean that decisions are being made without values; it merely means those values are being determined in a murky and indistinct way, hidden from scrutiny and often not obvious even to those involved in the decision process. Only by making trade-offs and values clear can we hope to improve use of the earth's scarce and limited resources.

The challenge of decision making across integrated systems

Low entropy (i.e. previously unused or raw) natural capital resources have an amazing diversity of potential uses. However, the more that capital is used the greater its entropy and the less available it becomes for alternative uses. In some cases this is a simple binary choice; using a soil resource to grow food often means that it cannot be simultaneously used to produce timber. However, often this relationship is more complex, so using water for intensive food production does not necessarily mean that it is not subsequently available for drinking – but does often mean that it has to be treated before this subsequent use. This interconnection of uses means that changing our use of natural capital for a certain specified end almost always has implications for other uses (consider again the use of land for agriculture and the myriad of impacts this has alongside food output). Any decision which ignores these other consequences is clearly flawed; whether it be that it either under-or overestimates the net effects, or indeed results in decisions which are wholly deleterious for society.

Unfortunately such incomplete analyses are commonplace. Some decision makers may have preconceived notions of what is important and focus upon those consequences rather than the wider picture. Often that is because the remit of the decision is constrained. So a Government department charged with increasing food security may well fail to adequately consider the wider environmental and societal impacts of its actions (sometimes even to the long term disadvantage of its own remit; e.g. promoting forms of intensification which boost short term food production only to compromise long term output). Within the private sector businesses typically focus upon those consequences of its investment decisions which improve profits for its owners and shareholders. This is not, in our opinion, morally reprehensible. Indeed in many legal contexts the management of a firm is legally obliged to operate in ways which benefit its owners. This does not mean that businesses are obligated to maximize short term profit irrespective of its wider social and environmental impacts. Indeed, in the UK, the Companies Act 2006 (H.M. Government, 2006) requires business directors to promote the success of the company with regard to a number of factors including the likely long-term consequences of a decision and the firm's impact on the community and the environment. Nevertheless, the obligation to enhance shareholder value is clear and this will result in a focus upon the output of goods which have market priced values, often at the expense of other non-market, unpriced goods. This in turn means that public regulators need to consider policy frameworks which align the profit incentives of businesses such that they also promote the interests of wider society, including environmental sustainability.

As suggested above, incomplete analyses of the consequences of natural capital use are far from the sole provenance of private businesses. The public sector has frequently implemented policies which ignore the wider environmental impacts of decisions. A classic example is provided by the EU Common Agricultural Policy (CAP). While this has been substantially revised and improved in recent years, its early operation focused almost exclusively on boosting the production of food without consideration of the environmental consequences of such actions. This was justified at the time with an appeal to the fear of food insecurity which was a major driver for the creation on the European Common Market in the post-war period. Indeed an argument that one objective supersedes all others is a common hallmark of many poor policy decisions impacting upon the environment. Such an argument is never optimal; poor policies impose unjustified and avoidable costs upon society and natural capital which always have to be addressed in the long term and are better avoided from the outset. The catalogue of policy reversals which characterise the history of the CAP provide an obvious example of unsustainable policies within limited focus (e.g. subsidies for hedgerow removal being superseded by subsidies for their replacement).

In summary then, the integrated system that is the natural environment requires that we consider all of the major consequences of decisions. Multiple effects are the common consequence of changes in natural capital and these need to be incorporated within decision making.

The challenge of decision making across non-commensurate natural metrics

If decision makers are interested in the overall impact which changes will have upon society then appraisals need to consider all of the impacts of an investment; not only the policy focus (e.g. boosting agricultural production) but also all consequent trade-offs ('externalities' such as water pollution), be they negative or positive. Failure to consider wider effects is common. Indeed by definition analyses which restrict assessment to a subset or single consequence of change cannot claim to be maximising (or, strictly speaking, even improving) wellbeing. Nevertheless, such 'cost-effectiveness analyses' (CEA) are commonplace. There is no doubt that such appraisals are potentially of use, for example if an agency is solely charged with say improving water quality then CEAs can significantly enhance the level of impact delivered by a given budget (e.g. the costs of reducing diffuse water pollution can be very substantially reduced using measures highlighted via CEA; see Fezzi et al., 2008, 2010; Hutchins et al., 2009). However, claims regarding the overall social value of interventions assessed in limited analyses have to be treated with caution.

A substantial challenge to comprehensive assessment of investments which affect natural capital is that impacts are often most naturally measured using an array of differing metrics. So, flood control is most obviously assessed in terms of risk per household, drinking water quality in mg/litre of pollutants, greenhouse gases in tonnes of carbon equivalent, recreation as the number of visits, and so on. These measures are typically non-commensurate; for example how many recreational visits should be given up to sequester an additional tonne of a given greenhouse gas? It is effectively impossible to trade-off ecosystem services without making them commensurate (indeed decision making which avoids comparability is essentially random as it either ignores certain trade-offs or makes decisions which only implicitly assigns values to them).

Given that the overall objective of natural capital investments is to improve wellbeing then the logical approach to commensurability is to assess the extent to which each trade-off contributes to wellbeing (either positively or negatively). By moving from a myriad of units to a single measure, and making sure that measure relates to wellbeing, we address the commensurability problem. Arguably one could use any unit to provide this comparability. But what is the best unit with which to assess changes in wellbeing? Ideally we would want a pure unit of wellbeing, or as economists term it, utility. Unfortunately there is no readily available 'utility-meter'. Therefore an alternative is to use a unit which people commonly use to express the wellbeing they obtain from the gain or loss of a good. This of course is not a challenge that is confined to natural capital related goods and throughout history society has solved the problem of how to exchange different goods through the medium of money.

Using money as a unit of wellbeing for making commensurate the multiple trade-offs associated with natural capital change has important benefits. A commonly claimed advantage is that decision makers are familiar with money; but this rather general assertion hides a more important truth. Remember that all investments, including those which concern natural capital and those that do not, face resource constraints. So, if those investments are being considered by the public sector, then the Government needs to ensure that the limited tax funds at its disposal are allocated wisely, in the way which will maximise wellbeing for the funds available. Society needs a robust natural capital base and high quality environment. However, it also needs a health service, education, transport infrastructure, employment, security, etc. All of these other requirements also draw upon the finite resources available to the Government. Furthermore, advances in economic assessment means that almost all of the benefits of these other demands upon public finance are now assessed in terms of economic values.

This is not to claim that money is the perfect common unit with which to express diverse benefits. Conversion problems abound, but these are even more challenging when other units are used. Indeed it would be more accurate to argue that money is simply the least-worst common unit available. The long term failure to assess the benefits of investing in the natural environment in monetary terms has coincided with long term over-use and degradation of natural capital. Is this pure spurious correlation, or at least in part causality? By insisting that environmental investments are assessed in a myriad of non-economic, non-commensurate terms, the wellbeing generated by such investments cannot be compared with that of the other spending a government undertakes. This has resulted in the environment being seen as a net cost yielding little obvious benefit. Certainly the case for increasing spending on the environment is difficult to make when expressed in such diverse and unfamiliar units. Ultimately failure to value the benefits of investments in improving natural capital leads to a decision maker having to compare an investment in the environment expressed through a diverse array of non-commensurate units with say an investment in building hospitals for which the economic benefits of lives saved have been assessed. Given this it is hardly surprising that public spending on the environment typically represents a tiny fraction of GDP.

The advantages of estimating environmental benefits and costs in terms of economic value has long been recognised and has resulted in a well-established research literature. While marketed goods are often valued with reference to their prices, a range of methods have been developed for valuing non-market goods (Freeman et al., 2014; Champ et al., 2017). These methods can be broadly divided into three categories as follows:

- Production function methods which examine of how changes in the environment and ecosystem services affect economic output, e.g. how changes in the climate affects agricultural production (Fezzi and Bateman, 2015);
- Revealed preference methods which infer individuals' preferences and hence values through observing behaviour, e.g. looking at the time and expenditure which visitors spend to reach preferred recreational sites (Herriges and Kling, 2008);
- Stated preference methods which use experiments or surveys to ask respondents to either directly state their willingness to pay for changes or to choose between alternative outcomes with differing costs, e.g. examining choices between different levels of water bill according to the quality of river water they offer (Metcalfe et al., 2012).

Non-market valuation methods are ideally suited to the estimation of the multiple values which can arise from changes to natural capital. So, for example, changes in land use can generate multiple ecosystem service impacts. So changes in flood risk can be valued by looking at expected monetised impacts upon households and infrastructure (Penning-Roswell and Green, 2000). Water quality impacts can be assessed by looking at the values generated or lost by changes in the water quality resulting from agriculture or industry (Moxey, 2012). Related impacts on recreation can be valued by looking at choices made by visitors across sites and relating these to the costs they incur to visit those sites (Herriges and Kling, 2008). If changes in recreational access can be shown to affect visitors health or life expectancy then this can be valued by examining people's willingness to pay for changes in health risk (Krupnick et al., 2002) although simpler estimates of health costs (which should not be confused with true values) can be obtained either by looking at impacts on production (Murphy and Topel, 2006) or the avoided costs of illness (Tarricone, 2006). Similarly if the land use change alters exposure to noise then this can be valued by examining the premiums people are prepared to pay for properties away from sources of noise such as roads and airports (Day et al., 2007). Furthermore, if the change in land use affects the emission or storage of greenhouse gases then these can be valued with reference to the damages expected to be generated by climate change (Tol, 2002) while the cost of abating those emissions through alternative means can also be assessed (Criqui et al., 1999; Kuik et al., 2009). Typically then a substantial majority of the benefits and costs of environmental change can be assessed using economic values, thereby facilitating their comparison. Note that, in all these cases, these are not the values postulated by economic experts, rather they are social values as reflected in individual behaviour.

Assessing impacts on wild species and biodiversity

While the majority of environmental costs and benefits can be robustly assessed using economic values, a few remain particularly challenging. Amongst these, the valuation of impacts on wild species and biodiversity raises particular problems.

Certain aspects of biodiversity value can defensibly be estimated in economic terms (Hanley et al., 2015; Pascual, et al., 2017). Indeed in some of these cases it is the natural science task of understanding the physical changes in wild species and their populations which is the principle challenge. So, for example, provided that we have a clear understanding of the relationships between wild species, plant pollination and crop production (a substantial research challenge spanning the natural and agronomic sciences), the monetisation of those changes in output via the market prices of those crops is relatively trivial (Losey and Vaughan, 2006; Melathopoulos et al., 2015; Breeze et al., 2016). Similarly we can look at the uplift in recreation values generated by biodiversity by examining how much further and often people are prepared to travel to experience particular aspects of biodiversity, such as viewing rare birds or hunting (USNCR, 1999; Kolstoe and Cameron, 2017). The additional costs incurred by those involved provide an input into revealed preference assessments of the value that wild species generate (Champ et al., 2017).

The examples cited above refer to the use values generated by biodiversity and derive values through observations of human behaviour. However, it is well established that biodiversity also generates non-use value; the value that individuals enjoy simply from the knowledge that wild species continue to exist and will be bequeathed to future generations (Kotchen and Reiling, 2000; Diafas et al., 2017). The lack of output effects or observable behaviour means that production function and revealed preference methods are not applicable to estimating non-use values. Arguably they may be inferred by examining direct payments for conserving wild species though donations, memberships of conservation groups and legacies (Pearce, 2007; Simpson, 2007; Atkinson et al., 2012). However, such approaches will at best provide poor under-estimates of true value (an expectation confirmed by the low values reported by such analyses), well out of synch with other measures of concern for the conservation of species.

In theory the non-use values associated with biodiversity can be directly estimated using stated preference methods such as contingent valuation and choice experiments. (Hanley et al., 2003; Christie et al., 2004; Morse-Jones et al., 2012). In practice such exercises face a number of challenges. One problem is that many studies have found the general public to have "low awareness and poor understanding" of what biodiversity means (Christie et al., 2006: p.305). Communicating such information to survey respondents is difficult given compelling evidence that information can alter preferences and values such that they are no longer representative of the social values researchers are seeking to estimate (Samples et al., 1986). Furthermore, unlike many other applications of stated preference techniques, studies seeking to estimate values related to conservation often cannot use payment scenarios in which the respondents are forced to make the payments they agreed to (unlike say studies which use water bills as 'payment vehicles' for delivering changes in ecosystem services). This violation of 'incentive compatibility' mean that respondents do not face a cost if they misrepresent their values (Bateman et al., 2002); an issue which challenges the robustness of resulting value estimates.

We therefore face a challenge. We have established that most of the ecosystem service impacts generated by environmental change can be robustly assessed using economic values – but not all. Simply omitting those benefits and costs from analysis would produce decisions which did not reflect society's preferences. Abandoning economic valuation would move us back to reliance upon the indistinct implicit values criticized previously. So how do we ensure preferences regarding non-monetised values are not ignored? In the case of biodiversity fortunately we have plenty of other evidence regarding preferences that we can bring into play.

Clear evidence regarding the ubiquity of non-use values for wildlife and concern regarding potential losses of such values are well established through repeated surveys. For example, the most recent UK *Public attitudes and behaviours towards the environment* survey (National Statistics, 2009) revealed that 91% of respondents agreed that 'there are many natural places that I may never visit but I am glad they exist' while 85% agreed with the statement that 'I do worry about the loss of species of animals and plants in the world' (*ibid.*). These strong social preferences are reflected in law. For example EU regulations on the protection of wildlife and biodiversity include directives on: species conservation (including limits to hunting and regulations on trading in wild species, European Parliament, 2009); habitat conservation (which includes designating areas of specific ecological need for species, European Parliament, 1992); and the protection of endemic species against invasive alien species (European Parliament, 2014). Although we can estimate the costs of delivering such regulations, this does not in any way equate to the value of biodiversity. However, this diverse survey and legal evidence regarding the strength of non-use preferences shows that people hold high values for avoiding the loss of wild species. This provides us with a simple yet effective way of incorporating this preference information into decision analyses by simply requiring that any potential change to natural capital should conform to a biodiversity objective to avoid any losses in wild species. By requiring that any proposed investment which contravenes this conservation objective is ruled out of consideration for funding we can ensure that individual's values regarding wild species are incorporated into decisions – even though we do not know (and indeed do not need to know) the non-use value of conserving wild species. This approach is applied in practice within our national level case study.

Of course we could impose a stricter requirement that, rather than just avoiding biodiversity losses, potential investments have to contribute towards the improvement of biodiversity. Preference data to support such an objective would be required but arguably is already available (National Statistics, 2009). Furthermore, as has been argued for a considerable period, alongside its direct use and non-use value, biodiversity supports a variety of ecosystem service related benefits which may be too complex and poorly understood to be adequately captured within most assessments (Mace, 2014; Mace et al., 2015; Turner and Daily, 2008; Bolt et al., 2017); an observation which mitigates towards a precautionary, standards based approach to both avoiding losses and indeed delivering gains in biodiversity (Bateman et al., 2011b; Harper, 2017). However, such an objective raises a number of practical questions regarding the desired level of improvement as this will inevitably involve some trade-offs. Rather than engage in such debates, which clearly require ecological inputs, in this paper we merely confine ourselves to proving the point that biodiversity can be defensibly integrated into a natural capital decision making approach without the necessity of resorting to dubious estimates of the economic value of the non-use benefits it provides.

Payment mechanisms: Uniting payers and providers of ecosystem services

As part of any investment analysis consideration needs to be given to who will provide a given change and who will fund this provision. Indeed the available 'payment mechanism' is an important element of the appraisal process. Figure 2 overviews the potential payment mechanisms available.

Figure 2: The Payer-Provider matrix for Environmental Goods.

		Provider (of goods)	
		Private	Public
Payer (for goods)	Private	Natural Capital markets: profitable environmental improvements (Business to business case study)	Corporate Social Responsibility projects
	Public	Subsidies to businesses (National and catchment level case studies)	Tax funded public provision

Considering Figure 2, the most common payment mechanism for the provision of non-market environmental goods is where the public sector provides funding while the private sector provide the goods in question, e.g. CAP subsidies are paid to farmers to provide conservation services. A common challenge for public funding schemes is that, while the natural environment, and hence its ecosystem services, vary markedly across locations, subsidies are often allocated in an untargeted manner, with flat rate payments to all areas being commonplace. While such an approach is easy to administer, it is highly inefficient. Using environmental modelling and economic valuation allows the targeting of support towards those areas or interventions which yield greater benefits. This ensures that funders, ultimately tax-payers, receive better value for money. It also means that the same level of resource generates enhanced environmental outcomes. The national and catchment level case studies presented below illustrate recent innovations in enhancing the efficiency of public sector funding schemes through targeting.

Further improvements in the efficiency and impact of funding can be delivered through the use of markets to allocate support payments. By creating market structures which induce competition between ecosystem service providers, the incentive for private firms to over-charge for their actions is reduced. So, rather than simply allocating support payments to providers, funders can announce the availability of funding and invite competitive tenders for the provision of ecosystem services. This should drive down costs and hence enhance the social benefits delivered by a given level of funding. Combining such 'natural capital markets' with resource targeting further improves resource efficiency and value for money.

Of course, from a public sector perspective, these mechanisms are further enhanced if it is the private sector which finances these initiatives. Corporate Social Responsibility investments now represent a substantial source of private sector funding for environment projects involving major multinational corporates. For example since 2012 Microsoft's global operations have been completely carbon neutral (Microsoft Corp, 2017), an initiative recently taken up by Google (Hölzle, 2016; Google, 2016). Most recently in 2017 the Mars Corporation launched a massive $1bn programme to achieve environmental sustainability of corporate operations within a generation (Mace, 2017). While such investments clearly represent short term costs to such companies, the social and reputational benefits generated by environmental improvements may well raise sales or generate price premiums (Bateman et al., 2015). Moving more in the direction of conventional profit bearing activities, many companies invest in areas which overtly yield a mix of both private and public benefits. For example, Häagen-Dazs (2017) has invested substantially in approaches to sustain honeybee populations, recognising both that they are of considerable non-use value to society and that they are vital to the ingredients supply chain of the ice cream manufacturer. Given these trends, our business to business case study considers the combined advantages of using environmental and economic valuation research, natural capital markets and private sector funding to deliver profitable environmental improvements

Spatial Scaling and Targeting

From a pure natural science perspective there is a good argument to say that there is no single perfect scale for decision making involving an ecological system. This situation is further complicated because of intersecting administrative levels and yet further boundaries around areas across which the economic benefits of ecosystem services arise (Bateman et al., 2006a). Scales and boundaries exist at multiple levels and their overlaps can cause conflicts in decision making. Water issues are often best dealt with at catchment level, but even these will often sit within wider basins. Administrative scales, such as regional and national level boundaries, often overlap natural scales; for example the River Severn supplies water to many areas in the English Midlands yet rises in Wales. Further boundaries also exist at the level of economic value, that is, a boundary lies where economic value is obtained and these can also overlap with the ecological and administrative boundaries. We have to recognise these boundaries, overlaps and conflicts when making decisions to try to do our best to delineate the spatial scale that is most suitable for the investment decision.

A further spatial issue concerns the degree to which policies are targeted. Untargeted policies effective ignore the natural variation in the environment; for example the UK Entry Level Stewardship scheme (Natural England, 2013), offers a standard level payment for environmental actions irrespective of their location. This in effect fails to link payments to outcomes in a manner thereby mitigating heavily against the potential efficiency, impact and hence value for money of subsidies. All three case studies consider the issue of spatial targeting and its relevance to environmental decision making.

Layout of subsequent sections

The rest of this chapter presents out three natural capital decision making case studies. Our initial national level case study examines alternative land-use futures to 2060. It highlights the advantages of valuing both the market and non-market effects of change and the importance of spatially targeted policies and. This example also illustrates the importance of understanding the drivers of change, thereby allowing decision makers to see where a shift in a particular policy is likely to result in changes in land-use, ecosystem services and values. The second, catchment level, case study examines the chain of drivers and knock on behavioural responses triggered by climate change and the resultant impacts on land-use and its various benefits. Our third and final, business to business, case study presents a novel approach to the funding of natural capital change showing how mutually beneficial business to business payments can be used to enhance private profits through the delivery of environmental improvements.

Public to private funding of natural capital improvements: National level decision making

I. Description of the problem

The National Ecosystem Assessment

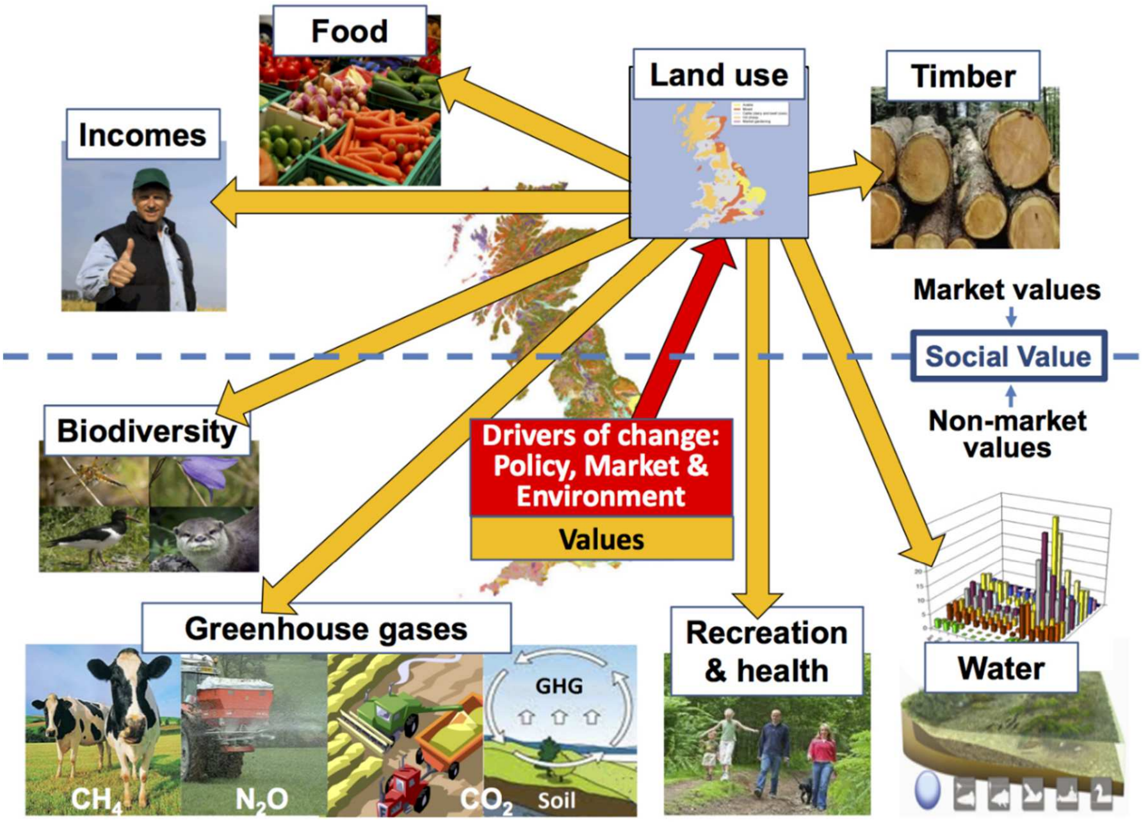
The Millennium Ecosystem Assessment (2005) highlighted the global problem of ecosystem service degradation at a global scale and urged action at all governmental levels to address this problem. The first major national level response to this challenge was provided by the UK through its National Ecosystem Assessment (NEA). This provided the first comprehensive assessment of the United Kingdom's ecosystems and included a systematic environmental and economic analysis of the benefits they generate (UK NEA, 2011; Bateman et al., 2011a, 2011d). The NEA sought to assess the consequences of natural capital use and land-use change and showed that over 30% of the services provided by the UK's natural environment are in decline. The UK Government responded quickly and positively to the challenge of the NEA, bringing forward new legislation to implement change and setting up the world's first Natural Capital Committee to provide it with independent advice (H.M. Government, 2011). This subsequently lead to the NCC recommendation for a 25 Year Environment Plan (25YEP) (NCC, 2015), a recommendation which was subsequently adopted by all of the major UK political parties and accepted by the Government. The NCC recently delivered detailed advice regarding this plan, including the harmonisation of agricultural policy within a wider environment and land use policy (NCC, 2017d), and the UK Government has agreed to consider this advice within its own 25YEP to be published in 2017-18 (Gove, 2017).

The data provided by the NEA (UK NEA, 2011) formed the basis of the models used in the assessment outlined in this case study which is based on work by Bateman et al. (2011a; 2013; 2016). A wide range of highly detailed, spatially referenced, time series, environmental data covering all of Great Britain was collected, ranging from soil characteristics (e.g. susceptibility to water logging), climate variables (e.g. temperature and rainfall) and land-use (e.g. agricultural output). This was complimented by similar spatially and temporally referenced data on market variables (e.g. prices and costs) and policy (e.g. incentives such as subsidies, and regulations such as land use constraints). The resultant analysis (ibid.) linked environmental, policy, market and other economic factors to examine both the market and non-market consequences and values generated by land use and changes thereto. The spatially sensitive nature of these analyses also demonstrated how future policy can be targeted to most efficiently allocated available resources to maximise their net benefits.

The Drivers of Change

Figure 3 illustrates the drivers of change, outcomes and principal resultant values associated with agricultural land-use in the UK and incorporated within the conceptual framework of the NEA (Mace et al., 2011). The NEA analyses sought to illustrate the potential advantages or costs of alternative future land use scenarios. Each analysis began from an econometric model of the drivers of land-use (Fezzi and Bateman, 2011). This model drew upon a long term (roughly 50 year) dataset covering the entirety of the country (with data on land-use, its drivers and consequences, held at a 2km square resolution or finer). A key element of the model was its specification to incorporate change into policy, market and environmental drivers. So, for example, in estimating land-use change over the long term (50 years into the future) the model allowed incorporation of predicted climate change into all assessments (Fezzi et al., 2014). With this incorporated into each analysis, the NEA set out to consider six policy scenarios, each in turn being considered under both a high and low assumption regarding future GHG emissions.

Figure 3: The Drivers of Change and Land-use Decisions.



The land-use predicted from each climate/policy change scenario was highly disaggregated, detailing the number of hectares devoted to each multiple land uses (and within agricultural areas more than a dozen farming land-uses) within each 2km square across the entirety of Great Britain. As each scenario posited differing climate, policy and market conditions so the analytical modelling predicted differing land use. Each predicted land-use served as the base data inputting to a series of interlinked ecosystem service impact and economic valuation models detailing the delivery of food production, the emission and sequestration of greenhouse gases (including CO_2, CH_4 and N_2O), expected numbers of open-access recreational visits, levels of urban greenspace amenity and a series of biodiversity metrics (in this analysis based upon wild bird-species diversity, chosen because of superior spatio-temporal data availability although more recently broadened to include a variety of mammals, insects and birds). Details for these various models and the derivation of outputs and values are given in a series of papers (Abson et al., 2014; Bateman et al., 2014a; Fezzi et al., 2014; Perino et al., 2014; Sen et al., 2014). The ability to spatially target policies using the outputs of this analysis represents a significant policy advantage of this approach (as discussed in detail in Bateman et al., 2014b).

II. Biophysical explanation of the relevant ecosystem services in the specific case

The different ecosystem services in the analyses were a mixture of market, non-market and cost-effectiveness valued services:

- Food output provides the key, market valued, ecosystem service determining approximately 75% of land-use in the UK, including cropland, grassland, mountain, moor and heathland environments. Values are obtained from market prices (Bateman et al., 2013).

- The sequestration of greenhouse gases (GHG) has a non-market value and includes the direct and indirect emissions from land-use and land management, annual flows of carbon from soils due to land-use changes, accumulations of carbon in terrestrial vegetative biomass and emissions. With certain exceptions (e.g. peatlands) GHG sequestration and storage is higher under woodland than agricultural environments and this is reflected in scenarios which see the transfer of land from farming to forestry. Values can be obtained through various routes including estimates of the expected damage of climate change, the cost of abating emissions, and the values of carbon traded in emission markets. This gives a range of values the implications of which are considered in Abson et al., (2014).
- Open-access recreational visits have a non-market value which varies across environments (e.g. mountains, coasts, forests, urban greenspaces, city centres etc.) and locations (with visitation declining with increasing distance from populations). The present study captures the difference in recreational values generated by different environments (our catchment case study further captures the impacts of changing quality within an environment by examining the impact of differing water quality at riverside recreational sites). Values were obtained through meta-analyses of prior preference based studies (Sen et al., 2014).
- Urban green space has a non-market value reflecting aesthetic, physical and mental health, neighbourhood, noise regulation and air pollution reduction benefits. Values were obtained through meta-analyses of prior preference based studies (Perino et al., 2014).
- Wild bird-species diversity was used to represent the biodiversity across the UK because they are high in the food chain and are often considered to be good indicators of wider ecosystem health (Gregory et al., 2005). As discussed previously, we argue that current estimates of biodiversity values, and in particular pure non-use existence value, are insufficiently robust. Following the reasoning set out previously we therefore impose a 'no-loss' constraint on the biodiversity consequences of any proposed future land use scenario, ruling out investments in areas which negatively impact upon our wild species metric (Bateman et al., 2011, 2013).

III. Identification of the beneficiaries

The same change can yield very differing consequences to different groups. Given this, decision analyses can yield widely varying conclusions depending upon the inclusivity of their remit. The national level remit of the present analysis examines various alternative scenarios for future land use and for each of these it attempts to quantify the resultant benefits and costs across all members of society. So we consider both the market and non-market net benefits to farmers, foresters, recreationalists, wildlife enthusiasts, etc., indeed all members of society. This allows the decision maker to make a comparative assessment of these scenarios and understand which provides the best value for money to society. Of course each scenario offers not only different levels of net benefit to society but also differing distributions of these benefits across groups. Within the present analysis we ignore these distributional issues and focus upon the overall benefits to society. Consideration of distributional effects for certain ecosystem services is provided in Bateman et al., (2011a) and Perino et al., (2014). However in summary the specific beneficiaries of alternative land uses include:

- Farmers: While climate change will deliver a negative impact on agricultural production in many areas of the world, the latitude and generally colder climate of the UK means that an increase in temperatures is likely to increase profits from agriculture in those areas which are not liable to drought (Fezzi et al., 2014; Fezzi and Bateman, 2015). However, this in turn is likely to negatively impact upon water quality and hence water supply customers as higher agricultural intensification leads to increased nutrient pollution in waterways necessitating higher treatment costs (Fezzi et al., 2015). Lower river water quality will also impact negatively upon freshwater biodiversity and river related recreational values (Bateman et al., 2016).

- Recreationalists: Open-access recreational sites have a benefit to individuals who visit them. The net benefit of sites declines with increasing distance from an individuals' home or outset point. This in turn highlights the importance of spatially targeting land-use decisions to obtain the greatest value from these ecosystem services.

- Urban residents: Urban greenspace value is reflected in local property and rental value, with that value generally decaying as distance increases (Day et al., 2007; Andrews et al., 2016). Typically increasing access to urban greenspace generates significant social benefits. However, the distribution of benefits can be uneven and can result in the gentrification of areas which has the potential to push poorer families out to less advantaged areas. Recently developed techniques such as Equilibrium Sorting Analyses seek to capture this effect and bring it into decision making (Binner and Day, 2015).

- Biodiversity beneficiaries: Improvements in wild-species diversity not only benefit the species being directly or indirectly (e.g. through food chains) conserved, but also any of those who value such improvements. These include those who hold use values (e.g. those who engage in hunting or fishing as well as those who enjoy viewing wild species) and those who hold non-use existence values. Biodiversity also indirectly delivers value through roles such as pollination and through diverse contributions to the maintenance of ecosystems.

- National and global beneficiaries: Both the conservation of biodiversity and the reduction of climate change through the sequestration of greenhouse gases through land-use change not only benefits the UK but has spill-over effects worldwide.

IV. Identification of the suppliers

As shown in Figure 2, ecosystem services can be supplied by either the public or private sector. The supplier with the most potential to supply ecosystem services in the UK are private agricultural landowners and farmers who control the large majority of land. The business to business case study looks at creating markets in the private sector as a means of increasing the ecosystem services provided by private landowners.

V. *Quid pro quo*

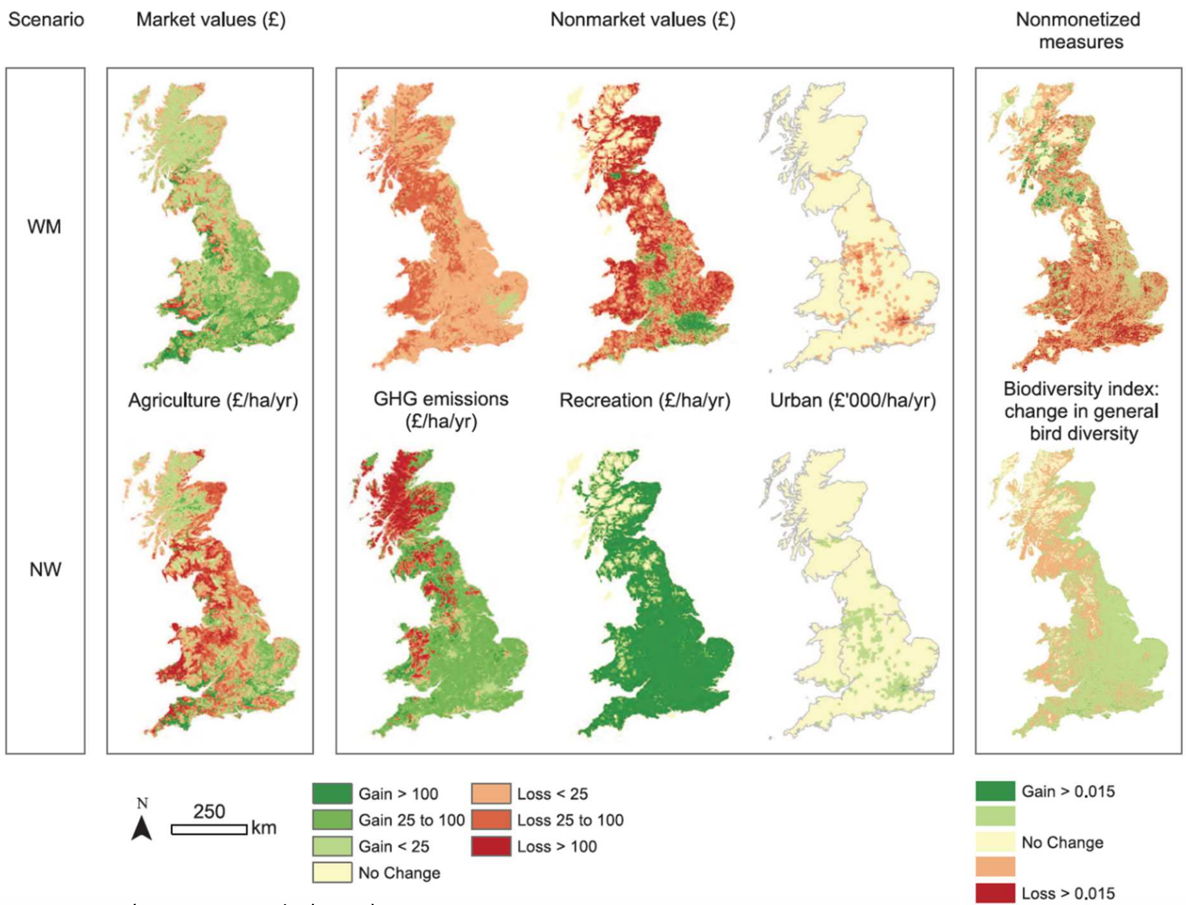
The analyses modelled land-use change for six different policy scenarios each of which ran from 2010-2060 to see the effects each would have on the land-use across Great Britain. Further details of all scenarios are given in Bateman et al. (2013), while for simplicity here we consider the two most extreme policy scenarios. The World Markets (WM) scenario prioritises economic growth by completely liberalising trade. Here trade barriers are dissolved, agricultural subsidies disappear and as a result farming moves towards large scale, intensive production methods. By contrast, the Nature@Work (NW) scenario's main priority is enhancing and maintaining the output of all ecosystem services. Adapting to climate change is also a priority (Bateman et al., 2011a: 1268). Within each scenario moving between high and low GHG emissions had relatively little impact on analytical results. This was mainly because most emission impacts are delayed towards the latter part of the assessment period such that a shift from high to low emission only changes values by 0.05% in the WM scenario and 3% in the NW scenario. However, within each emission level, shifts between the two policy scenarios generated highly significant changes in land-use, ecosystem services and resultant values.

Results: Considering market goods alone; ignoring non-market impacts

While it captures only a single dimension of impact, solely focusing upon market priced goods indicates values which are often given primacy in policy decisions (often due to a failure to adequately consider many other effects of change). The WM scenario sees an agricultural value increase of £1.03bn p.a. because of a shift towards more intensive production. Conversely, the extensive approach of the NW scenario sees agricultural values decline by £0.13bn as farmland is converted for the generation of urban-fringe greenspace and recreation. These differences are illustrated in the first column of the maps shown in Figure 4.

Considering the agricultural values shown in Figure 4 an initial observation is the strong spatial heterogeneity of changes in value under each scenario, highlighting the clear advantages of adopting a spatially targeted approach to any policy affecting the natural environment. However, comparing across scenarios reveals further major changes in value from the baseline. This restricted analysis highlights the very clear result that if land-use decisions are based solely upon the production of market priced goods then a reduction in environmental regulations and intensification of pressure upon the land, as proposed under the WM scenario, almost always appears justified while the environmental restrictions imposed by the NW appear to generate losses across large areas.

Figure 4: The rationale for valuing ecosystems services: Impacts of two policy scenarios in Great Britain (World Markets (WM) scenario prioritises agricultural production; Nature@Work (NW) scenario seeks to enhance the overall value of all ecosystem services).



Source: (Bateman et al., (2013)

Results: Considering market and non-market goods

While our analysis of the impacts of these scenarios upon market goods favours the WM scenario, turning to consider the impacts upon non-market goods we find that the NW scenario which consistently yields preferable outcomes. GHG emission values in the WM scenario are negative in nearly all areas. In contrast, under the NW scenario most areas see benefits in terms of increased carbon storage, the exception being upland areas dominated by fragile peatlands which are vulnerable to both intensive agriculture (WM) and increasing forestry (NW). This immediately suggests that neither scenario may be optimal, an issue considered in subsequent work (Bateman et al., 2014b). Nevertheless the NW scenario is clearly preferable to WM in most areas. Considering recreation impacts, the WM scenario sees losses in visitor values in almost all areas across the country while the NW scenario sees positive changes. Similar results are felt in urban greenspace values, where the changes are consistently negative in the WM scenario, but consistently positive in the NW scenario. As discussed previously, the lack of robust valuation methods means that biodiversity impacts are not monetised in this analysis. Nevertheless our biodiversity metric clearly shows that the WM scenario results in major declines in biodiversity across large swathes of the country. In contrast the NW scenario generates improvements across the lowlands which constitute the majority of the country although the picture in the uplands is more mixed with insignificant or weakly negative effects. Again this suggests that an optimal solution would combine elements of multiple policies (including others not considered within the two scenarios selected here but covered by the full gamut considered in Bateman et al., 2013). However, what is very clear from even a cursory inspection of Figure 4 is that simple reliance upon a subset of goods, most typically just those with market prices, can be a highly misleading heuristic for social decision making.

Table 1 quantifies these effects and highlights the stark difference between these scenarios in terms of their contrasting impacts on market and non-market goods. In sum, what we see is that the WM scenario increases the production of marketed agricultural output but at the cost of significant declines in all other ecosystem services which strongly outweigh the value of agricultural gains; in short this scenario lowers overall social value very substantially. In contrast the NW scenario reverses this pattern, causing a relatively modest reduction in agricultural production in return for very substantial increases in all other non-market ecosystem service related goods and a correspondingly major increase in overall social value. This disparity is further reinforced when we consider our non-monetised biodiversity measures which shows a decline under the WM scenario but an increase as a result of the NW scenario. If we applied our rule that any decision which lowers biodiversity in an area is ruled ineligible for that area then at a national level the WM scenario is unacceptable. Even at a local level then the NW scenario would be ruled out in some areas.

Table 1: The Effects of Policy Scenarios on Ecosystems in Great Britain.

Scenario	Market agricultural output values	Non-market GHG emissions	Non-market recreation	Non-market urban greenspace	Total monetised values	Biodiversity
WM	**1030**	**-440**	**-1180**	**-18400**	**-18990**	**-**
NW	**-130**	**230**	**13060**	**4760**	**17920**	**+**

Notes: All values are given in £ millions p.a. in real (inflation adjusted) from the baseline year (2010) calculated across GB. Positive (negative) values indicate net gains (losses). Details of valuation methods given in Bateman et al., (2014a). Biodiversity impacts are assessed through a widely applied measure of bird diversity across all species. Detailed classification given in *ibid.* and simplified in Table 1 so that + (-) denotes an aggregate national increase (decrease) in biodiversity (with local variation indicated in Figure 4). Scenarios use high GHG emissions variant; low emission results reported in *ibid.*

Source: Adapted from Bateman et al., (2014a)

Results: Summary

The analysis presented here attempts to bring together the multiple effects of land use change. In so doing some complexities are avoided such as the possibilities of non-linearities, thresholds and feedback effects. Clearly future research should try to incorporate these, but not at the cost of ignoring these wider multiple effects of change.

Accepting these caveats, when looking at alternative land-use decisions for Great Britain, examining only the agricultural values suggests that it is optimal to more intensely use the land and increase farm output by as much as possible. However, as soon as the analysis is extended to incorporate the non-market values of ecosystem services this result is reversed and an approach which seeks a modest reduction in agricultural intensity proves beneficial. Spatial targeting of these land-use decisions can also greatly increase their efficiency and raise the value for money of public spending in this area. When we look at wild bird-species diversity, increasing agricultural production without increasing conservation areas causes decreases in the levels of biodiversity (note that a further policy to increase yields in selected areas need not be deleterious for other ecosystem services provided that it is accompanied by offsetting policies to increase ecosystem services in other areas; see Lamb et al., 2016). Interestingly, the cost of implementing an objective to stop a reduction in the levels of wild bird-species diversity is relatively small in comparison to the high impact it has on the levels of biodiversity (see Bateman 2013 for details).

Considering the particular results of this analysis, the WM scenario fails both in terms of its cumulative contribution to social value and its overall impact upon our biodiversity objective. In contrast the NW increases social value substantially and provides an overall improvement to biodiversity. However in this latter respect a targeted approach would avoid biodiversity losses in local areas by adopting alternative policies from those locations (Bateman 2013). Adopting such a targeted optimisation approach can further enhance decision making (Bateman et al., 2014b).

VI. Mechanism for transfer of value

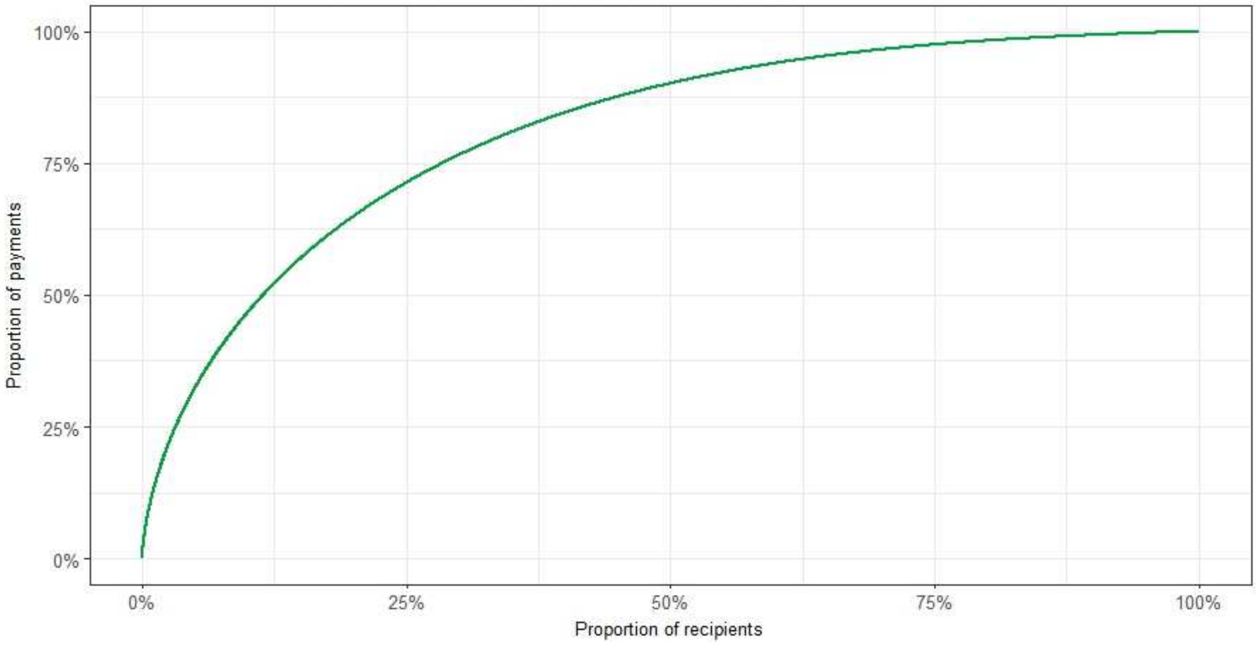
The most common current method for transferring farming related values in the UK is via Government funded subsidies. There is already a massive intervention in agriculture in the UK with regard to subsidies paid through the EU Common Agricultural Policy (CAP). As detailed in Table 2, these currently make up over 50% of UK farmers' income. However there are major arguments in favour of substantial reform of these payments (NCC, 2017). The present approach towards agricultural subsidies makes payments on a per hectare basis. While this might be argued as favouring most efficient producers, farm size is a poor proxy for such measures and from a political economy perspective the distributional consequences of such an approach, which results in almost three quarters of subsidies flowing to one quarter of farms (see Figure 5), including some of the richest in the country (Greenpeace, 2016), are difficult to defend. From an environmental perspective, these payments are unconnected to the degree of ecosystem service value generated and therefore offer relatively low value for money to taxpayers. Given this, the potential for reform seems very strong.

Table 2: UK Government subsidies to Agriculture 2011-2016

UK (£billion)	2011	2012	2013	2014	2015	2016
Farm income	5.434	4.895	5.585	5.297	3.903	3.61
Public subsidies*	2.805	2.6	2.691	2.337	2.176	2.568
Subsidies* as % income	51.6%	53.1%	48.2%	44.1%	55.8%	71.1%

Note: * = Only includes direct income payments; subsidies are even larger if indirect support (e.g. fuel subsidies) are included.
Source: National Statistics (2017)

Figure 5: Distribution of UK Government subsidies across farms 2016



Notes: Graph shows distribution of subsidies paid under the basic payment scheme (England)
Source: Defra (2017b)

Our analysis illustrates the potential for major improvements in social value and the efficiency of public funding through the spatial targeting of subsidies. By mapping which areas offer superior value for money for agriculture and which would be best turned over to the production of other ecosystem services and targeting funding accordingly the overall efficiency of subsidies can be greatly enhanced. From a policy point of view, the adoption of such spatial targeting offers a major and potentially costless increase in taxpayer value for money simply by redirecting the current level of subsidy according to the benefits it generates. These gains could be further elevated by adopting a simple 'public money for public goods' approach. Here the subsidy of goods which are subsequently paid for in markets is withdrawn. At present there is an argument that the UK population pays for its domestically produced food twice; once as taxpayers subsidising production and a second time as consumers purchasing food from shops. Markets convey demand for private goods, such as food, very efficiently, avoiding overproduction and waste. However, markets do not exist for public goods such as a high quality environment, clean rivers and air, wildlife habitat, open-access recreation, etc. Farms can play a very major role in providing these goods but farmers need to be compensated for any loss of agricultural production and hence income that this would entail. Targeted reallocation of existing subsidies would ensure maintenance of overall agricultural incomes and substantially boost environmental quality and ecosystem service benefits.

Within the UK the reallocation of existing levels of subsidy using such principles would also generate substantial improvements in terms of income distribution amongst farming households. The present system of allocating subsidy payments on a per hectare basis heavily favours much larger and typically richer farms. Reallocating subsidies would almost inevitably improve the distributional inequity illustrated in Figure 5.

The UK is currently in the process of leaving the EU and with it the CAP. Current subsidies will be maintained until 2020 but Brexit provides a unique opportunity to redefine agricultural policy and use contemporary evidence to deliver superior value for money to taxpayers through major improvements in the natural environment while simultaneously enhancing the livelihoods of the average farmer. Hopefully this opportunity will be seized.

VII. Monitoring and verification

As highlighted in Figure 2, Payments for Ecosystem Services (PES) bring together payers for and providers of valued ecosystem services. PES can involve both private and public providers of goods, but in this case study we consider payments from the public to private sector (in contrast to our final case study which considers PES schemes both paid for and provided by the private sector). The payments which are made in exchange for various ecosystem services are awarded, in this instance, on the inputs the provider makes to the service, i.e. on actions rather than their effect. For example, a farmer may be paid for reducing fertiliser applications on the assumption that this might reduce eutrophication in local rivers (the subject of our subsequent case study) but there is no linkage of that payment to the monitoring of river water. This means that farmers have no incentive to vary their actions away from those prescribed and payments may be inefficient in delivering the benefits they are intended to generate.

One important PES, the UK Countryside Stewardship Scheme (CSS), opened for the first time in 1991 covering only 25,000 hectares. By the time it was paused in 2004 the scheme covered over 530,000 hectares and its payments to farmers were in excess of £52million p.a. (DEFRA, 2006). The scheme was aimed at sustaining the beauty and diversity of the rural landscape and offered incentive payments to farmers for the preservation and provision of wildlife habitat. It made payments to farmers for arable land reversion, establishing and maintaining grasslands, as well as managing and preserving footpaths and bridleways (ibid: p.770). Participating farmers had to sign a contract which usually lasted 10 years and required them to maintain certain land-uses for that time. They were monitored through a series of checks including farm visits to see if the farmers were in breach of their contracts and if they were not providing the ecosystem service their contract was terminated and fines were sometimes issued. The payment authority set an objective to inspect 10% of the farms every year, but the actual inspection rate never reached this.

The CSS is now open again for applications (Forestry Commission and Natural England, 2017) and improvements have been made to the scheme since its conception and different levels of stewardship are available which differentiate between farmers' abilities. Currently a mid-tier stewardship offers 5 year contracts for environmental improvements in the wider countryside, such as reducing diffuse water pollution. The higher-ties focuses on environmental sights of importance which require more complex managements, such as commons and woodlands. Importantly monitoring approaches have also improved with farm visits now being supplemented by aerial photography from planes, the use of drone technology and increasingly data from earth observation satellites.

While improved monitoring of actions is welcome, further improvements could be achieved through a move towards payments by results. In some cases this may be reasonably straightforward to implement, for example the quality of water running off a field is something that, in some respects (e.g. pesticide or fertiliser content) a farmer can influence. However, other payment by results schemes are more difficult to implement. For example, with respect to wildlife, a farmer may provide food sources for birds to thrive, yet the bird roosts on another farmer's land. Questions then arise over who should be paid for such services. Despite this, payment by results has important advantages over the input based approach (Gibbons et al., 2011; Osbeck et al., 2012):

- Outcome based approaches incentivise land-use that will produce the best environmental result.
- It allows farmers more flexibility in the management of their land because restrictions imposed by the input based system are removed. Appropriately designed this could lead to a greater uptake of participants in the scheme and greater improvements if targeted appropriately on expected environmental outputs.

- Farmers are permitted to innovate and incorporate their existing knowledge into environmental provision which can lead to a greater efficiency of production. Extensions of this knowledge could be delivered through training schemes.

Improvements in monitoring the outcome of these PES could also be made to help reduce costs and help ensure that outcomes are met by farmers. The monitoring of some ecosystem services is easier to tie down to particular locations than others. As mentioned above, biodiversity is difficult to monitor because animals move from place to place, but services which are location specific lend themselves to outcome based monitoring. Monitoring water quality in rivers (see subsequent case studies) can be used to measure the outcome of reduced agricultural production, given that the diffuse pollution from farmland negatively affects water quality. The development of cheap monitoring equipment could be used to help monitor the water quality on farms in real time to assess the levels of nitrates and phosphates in the waterways (Schäfer et al., 2008). The development of this monitoring equipment could also be part-funded by the water companies who also benefit from the cleaner water. Importantly, monitoring which ties results to individual farms could radically improve the ability of decision makers to design PES schemes which incentivise results rather than merely actions.

The ongoing incorporation of drone based information, earth observation satellite data, and other 'big data' systems to monitor farmland-uses and certain consequences (e.g. tree planting and growth) offer further considerable potential for enhancing PES schemes.

VIII. Effectiveness

Effectiveness of the Changes if Implemented

The scenario analyses reported above illustrate the major impact hat policy can have on social values and the gains or losses that can arise from alternative decisions. While the WM scenario could generate losses of almost £19billion and major declines in biodiversity, the NW scenario offers a monetised gain of almost £18billion plus substantial improvements in biodiversity (additional tailoring of policies to different areas could further enhance these gains; see Bateman et al., 2013, 2014b). Implementing the NW scenario would deliver improvements in the all of the non-market goods considered in this analyses: with reductions in GHG emissions, increases in recreational areas and increases in urban greenspace benefits alongside biodiversity enhancements. The only area of society which experiences decreases is the agricultural production sector, but the total value gains to society could be used to compensate these losses to retain substantial overall net gains.

Effectiveness of the Modelling Carried Out

Fundamentally the most important finding of this research is that methods now exist to unite the natural sciences with economic assessments so that decision makers can be informed about the changes in value that will arise through different policy decisions. This allows policy makers to consider multiple land-use decisions in detail and select those options which yield the most desirable outcome. There are however a few limitations and challenges that need to be faced, these are:

- Simplifications had to be made because of uncertainties in predicting the future influence of certain drivers of change. Technological change represents this problem well, because advancements often do not follow a linear path and technological breakthroughs can revolutionise industries in unforeseen ways. Assumptions therefore have to be made. Here the modelling assumes an extension of prior rates of technological growth.

- A challenge that would need to be addressed if a spatially targeted approach towards decision making was adopted, is how it would coexist with the spatial insensitivity of the CAP during any transition period. Brexit offers an opportunity to renegotiate the subsidies made to farmers and build in spatially sensitive payments into future contracts to address this issue.

- A further challenge is how to efficiently target payments when the costs of delivering these services differ across land managers but are unknown to the funding authority. The business to business case study presented subsequently addresses this issue by implementing a new approach to contracting between farmers and funding authorities which incentivises farmers to reveal their true costs.
- Future analysis should also incorporate uncertainty into its modelling given that some policy effects are more certain than others, e.g. the modelling of water quality always includes some error regarding measurements, however assessments of the degree of error can be both made and incorporated into analysis.

Despite these caveats, the model remains an effective tool for policy makers. The results presented here were central to the UK NEA which in turn provided the principle empirical input to the UK Government's Natural Environment White Paper (H.M. Government, 2011) and subsequent policy action leading up to the current formation of a 25 Year Environment Plan. Following on from the work described above and a subsequent scoping exercise (Binner et al., 2017), the authors are currently contracted by the UK Government to develop the modelling framework into a natural capital decision support system; the Natural Environment Valuation Online (NEVO) tool. Early elements of this system are already freely available online, notably the Outdoor Recreation Valuation (ORVal) tool (Day et al., 2017), which was cited within a recent BBC report as the sole publically available output of the Governments 25 Year Environment Plan to date (Harrabin, 2017).

IX. Key Lessons learned

This analysis has built on our understanding of land-use decision making in a number of ways. Specifically, when making decisions about land-use change it is important to:

- Carefully determine the spatial extent of the area for analysis taking into account ecological, economic, policy and administrative jurisdictions.
- Understand the drivers of change; Policy, Market and Environment variables. This is important so that decision makers can understand which levers of change are available to them, and which are exogenously determined.
- Capture all of the major effects that changes in land-use can have upon ecosystem services; without this any attempt to estimate the overall consequences of policy change will be limited and can be potentially misleading.
- Make sure the different units used to measure changes in ecosystem services are commensurable. This is most readily achieved by expressing changes in both market and non-market goods in monetary economic values.
- While the large majority of values can be robustly assessed in monetary terms, some are not currently amenable to robust valuation. Biodiversity is a particular challenge here and we argue that generally agreed objectives (drawn in part from legislation but also from other preference measures and informed by ecological knowledge) should be incorporated into analyses by identifying and rejecting investments which violate those objectives. The costs of securing these objectives can be estimated and the effectiveness of options for their delivery assessed, but such costs should not be interpreted as indicators of the value of delivering sustainable biodiversity or other objectives.
- Account for temporal variation by ensuring that changes in drivers (for example those arising from the effects of climate change) are incorporated into analyses.
- Account for spatial variation across the study area, given the heterogeneity of land characteristics and responsiveness to change, to permit the targeting of policies.

- Allow for the modelling of different policy scenarios to see the different outcomes that can occur from varying the inputs. This is one of the most useful tools for policy makers to use. Ideally one would move from pure reliance upon scenarios towards the refinement of optimal policies given resource constraints and objectives (Bateman et al., 2014b).

This study is part of a series of inputs to the policy process which resulted in a new White Paper (H.M. Government, 2011) and is at the forefront of research into ecosystem services and land-use decisions within the UK. The Government is taking these findings very seriously and together with the NCC is moving towards a 25 year plan to improve the environment.

Public to private funding of natural capital improvements: Catchment level decision making

I. Description of the problem

While national level decision making is vital for directing overall and longer term strategy, most practical environmental management decisions are made at a more restricted and local scale. Nevertheless, the need for environmental coherence in decisions remains and in recognition of this there has been a noticeable movement towards consideration of catchments as an important decision making unit (see, for example, The Water Framework Directive (WFD), European Parliament, 2000; Bateman et al., 2006b, 2016). Irrespective of the scale however, the key principle for decision making is that all of the major effects of a proposed change should be incorporated into analyses, including relevant overspill benefits and costs arising outside the immediate locus of activity.

This section considers improvements to conventional public to private PES decisions at the catchment level. This is illustrated by analysis of the chain of related effects that climate change is likely to have upon a chosen catchment, in this case the Aire catchment in Yorkshire, UK. The increase in average temperature generated by climate change induces land-use responses as farmers exploit the potential for increased agricultural intensification. This in turn generates higher food production but this needs to be balanced against the impacts of induced land-use change in terms of increased nutrient application, higher river pollution and lower ecological quality. This then generates a non-market externality with relation to recreational visits to the river catchment. The problem is that if we just consider the direct market priced effects of land-use change, which is an increase in agricultural production, it appears that the temperature rise would increase the value to society. Extending our analysis to include induced indirect effects on water quality and recreational value provides a richer picture of the overall balance of benefits and costs.

II. Biophysical explanation of the relevant ecosystem services

The 86,000 ha catchment of the River Aire encompasses highly heterogeneous land-uses, water qualities and socioeconomic characteristics. The upstream western half of the catchment has low population density with its mainly upland landscape being dominated by rough grazing and pastoral agriculture. The downstream eastern half of the catchment has some mixed and arable farmland but is dominated by high density urban areas which are themselves a major determinant of river ecology and the major source of recreational visitors.

With rural land-use ranging from extensive pastoral to intensive arable farming across the catchment, the Aire is subject to significant levels of diffuse pollution including high nutrient loadings. Bateman et al. (2016) modelled the impacts of this pollution upon river ecological status. Farm effects upon water quality were isolated through a modelling exercise which controlled for the positive associations of eutrophication with temperature, and negative correlations with higher rainfall and river flow. Allowing for these effects revealed the arable and root crops were strongly and positively associated with high levels of eutrophication which also rose in areas of high stocking density. Conversely, extensive grassland systems were associated with less eutrophied waterways.

While much of the above analysis was conducted using secondary data sources, recreational behaviour and associated values were modelled using primary, large sample survey data to which revealed preference travel cost methods (Champ et al., 2017) were applied. However, recent advances in data availability through the official UK Monitor of Engagement with the Natural Environment (MENE) (Natural England, 2017) now allow such methods to be applied across the country. In conducting this analysis particular care was taken to avoid potential sources of bias. For example, the availability of recreational substitutes (including other rivers in and outside the catchment area and other types of recreational resource) were incorporated as were variations in the road infrastructure and the spatial distribution and socioeconomic characteristics of potential visitors. These analytic capabilities are now incorporated within the online ORVal model discussed previously.

III. Identification of the beneficiaries

The general public benefit from the river in two ways. First, they have a use value associated with river related recreation. This includes any activities done in and around the river e.g. walking along the riverbank, nature watching, canoeing, fishing, swimming, walking, cycling, running, etc. Second, they have a non-use value associated with the biodiversity levels in the river catchment quite separate from their direct enjoyment of wild species. Both recreational use value and non-use values were estimated using revealed and stated preference methods respectively. However, for ease of exposition, the present analysis solely focuses on changes in use value which should therefore be viewed as a lower bound on the total value of changes in water quality.

Water quality improvements also benefit water companies through reductions in treatment costs. The business to business case study explores how these latter private sector benefits can be assessed and brought into economic analyses and PES implementation.

IV. Identification of the suppliers

Farmers and landowners in the river catchment exert a major influence over river water quality. Climate change is predicted to induce increased intensification of agricultural production (Fezzi et al., 2014; Bateman et al., 2013) and hence greater applications of inputs such as fertiliser. This will in turn reduce river water quality through diffuse pollution. Conversely, farmers could maintain or even improve water quality by adopting more extensive production methods. As such, farmers are the potential suppliers of improved water quality. This in turn would avoid potential losses of recreational value and other associated ecosystem services. However, this would incur income reductions for those farmers. A key objective of this study was to calculate the forgone income and hence compensation required by farmers who avoid a move towards higher input extensive agriculture and compare this cost to the likely benefits generated by such action. In the subsequent case study we address the issue of how any such compensation should be determined at an individual farm level and the extent to which those in the private sector, who would benefit from lower levels of water pollution (in this case private water companies), might have a profit incentive to fund such compensation.

V. *Quid pro quo*

As per our wider methodology, the study starts off by considering the drivers of change. In this case, we hold the market drivers constant so prices just track general inflation levels and remain constant in real terms. But we note expected changes in the environment driver (climate change) and simulate this by assuming a 1°C increase in temperature across the region and some shifts in the seasonal pattern of rainfall. Subsequently, as discussed below, we also consider the effectiveness of policy drivers designed to counteract the negative consequences of climate change.

While global climate change will place greater stress on the world agricultural system, within certain temperate countries such as the UK, increases in temperature will actually result in elevated potential for agricultural production and profit (Bateman et al. 2013, Fezzi et al. 2014). Specifically, within the case study area, climate change will result in increased suitability for arable production. As this is associated with higher profits than many grassland systems, our analysis suggests that farmers will shift away from livestock production and towards cropping regimes. This will result in a greater use of the farm inputs associated with arable production; notably fertilizers. This, in turn, will increase the levels of diffuse pollution into waterways and rivers, resulting in elevated nutrient levels within freshwater environments. Therefore, the integrated suite of models employed to analyse this chain of events (Bateman et al., 2016) predicts a decline in the future water and ecological quality in catchment waterways which arises both from the direct effect of the water temperature increasing and from the (larger) indirect effect of the increased run off of nutrients into waterways from the climate induced shift towards arable farming.

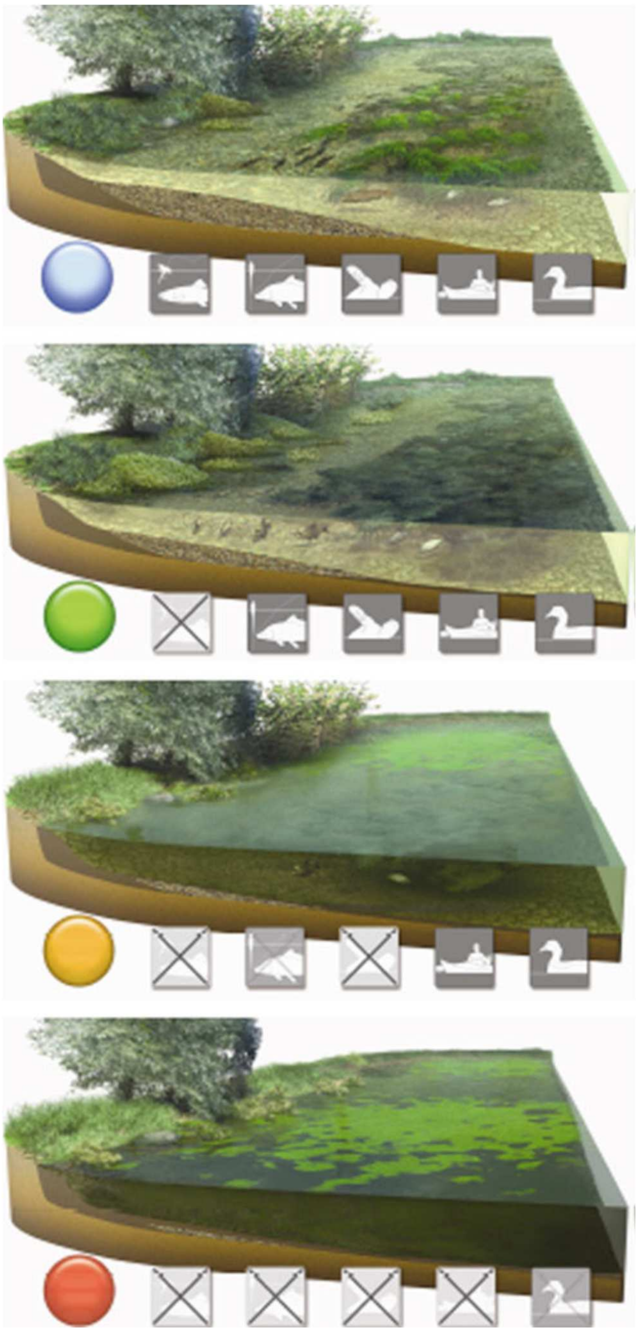
This decline in the ecological quality of the river not only reduces associated non-use values, it also generates a potentially major non-market externality in terms of the impact it has upon the recreational value of visiting the river. Given our concerns about the robustness of methods for estimating non-use values (Bateman et al., 2011b) we focus on the latter recreational benefits, in effect examining whether these alone are sufficient to justify compensating farmers to avoid this chain of land-use consequences. To estimate this value we need to understand the link between the ecological quality of rivers and the recreational behaviour and associated values of visitors. A large sample survey of households within a 70km radius was undertaken to estimate the recreational impact of changes in ecological quality. Such spatially distributed sampling allows us to take into account a decay in recreational values with respect to increasing distance of households from the affected river.

Testing for pollution levels

As a metric of the ecological impact of diffuse pollution in the river, an analysis of expected impact on chlorophyll-a levels was undertaken. Chlorophyll-a is used as an indicator of water quality in both natural and social science research because it identifies the risk of eutrophication of aquatic ecosystems. Eutrophication results in less dissolved oxygen in the water which in turn generates negative impacts on the native flora and fauna of those rivers. As a highly visible marker of eutrophication, chlorophyll-a is directly perceived by the general public as a measure of water quality. This reduction in water quality is associated with a decline in recreational quality, visits and values.

For the purposes of this discussion and to link the level of chlorophyll-a concentrations in the river to the perceived water quality affecting recreation, we use the ‘water quality ladder’ (WQL) illustrated in Figure 6. This links directly to UK official guidance on water quality (UKTAG, 2008) and identifies four broad classes of water quality which are proposed to be perceptually distinct; pristine (which the WQL denotes using a blue symbol), good (green), mixed (yellow) and poor (red, although there are no rivers of this quality in the case study area).

Figure 6: Water quality ladder (WQL) used in River Aire study



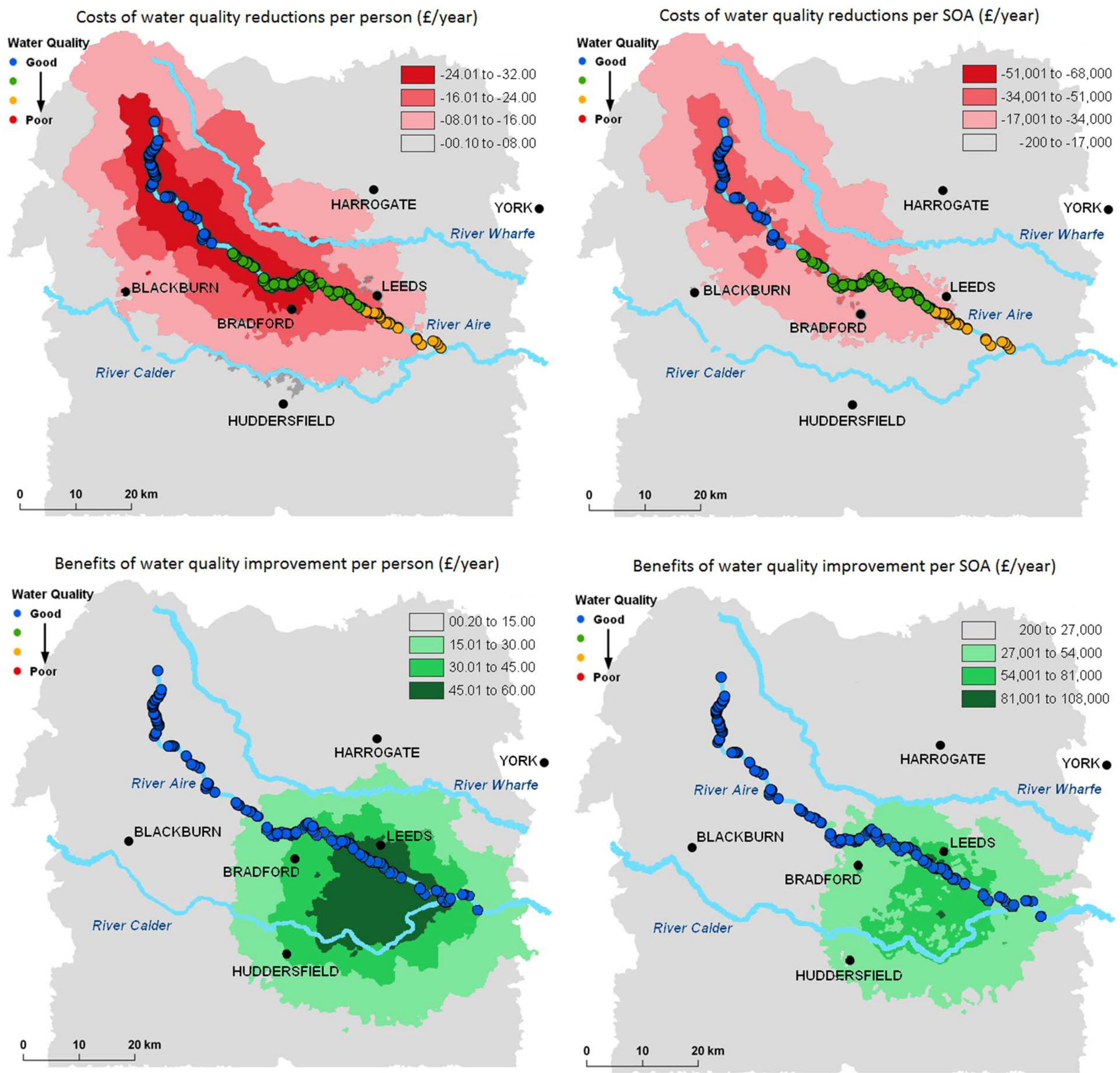
Source: Bateman et al., (2011c), copyright protected

Recreational value consequences of climate change driven reductions in water quality in the River Aire

The coloured dots in the maps shown in the upper row of Figure 7 uses the WQL to illustrate current water quality at recreational access sites along the River Aire. As can be seen, water quality is currently predicted as pristine for the upstream stretch in the west of the case study area. The direct and indirect effects of the 1°C temperature increase would cause these pristine sites to decline to 'good' water quality status as defined above (i.e. the sites currently coloured blue would change to green). Feeding this change into our revealed preference recreational value model allows us to predict the consequent losses to visitors. The coloured contours of the upper row maps indicate the level of recreational value losses and their spatial distribution. As expected, these losses are concentrated around the areas where the reduction in water quality occurs. Losses are mapped on both a per person (left hand panel) and per area (right hand panel) basis with the latter aggregated to Census Super Output Areas (SOA). On average the recreational value losses caused by climate change were £10.44 per person per annum across affected households (details in Bateman et al., 2016).

Figure 7: Upper row: The spatial distribution of per person (left hand panel) and SOA aggregate (right hand panel) losses in recreation value as a result of climate change.

Lower row: The spatial distribution of per person (left hand panel) and SOA aggregate (right hand panel) gains in recreation value under the WFD water quality improvement policy.



Notes:

- Colours represent water quality at sites: blue=Pristine; green=Good; yellow=Mixed.
- Upper row: Water quality at present; note that sites currently with pristine quality (blue) will decline to good quality (green) as a result of climate change.
- Lower row: Water quality following implementation of a policy to offset the impact of climate change. Note that this also results in an improvement relative to current water quality (as per the WFD objectives).
- Note that both the current water quality (upper row) and that expected as a result of climate change are predicted from the model of chlorophyll-a reported in Bateman et al., (2016), whereas water quality under the policy response scenario (lower row) is assumed to attain the objectives of the WFD.

Source: Upper row Bateman et al., (2016); Lower row, new calculations.

Policy analysis: addressing direct and indirect climate change impacts

The losses generated by climate change, both directly in terms of higher temperatures and indirectly through induced land-use change (Fezzi et al., 2015), can be mitigated to a greater or smaller extent depending upon the policy actions that are taken. Indeed the UK is a signatory to the EU Water Framework Directive (WFD) (European Commission, 2000) which requires EU member states to avoid reductions in water quality and work towards delivering good ecological status at all river sites. Using the objectives of the WFD as a policy guide we extend our analysis to incorporate the effects of implementing a policy, not only to avoid climate induced losses of water quality, but indeed to improve the River Aire to the pristine ecological standards required under the WFD.

The lower row of Figure 7 illustrates the consequences of bringing the water quality up to pristine standards at all recreational access sites (accordingly coloured blue as per the WQL). Using our revealed preference recreational model we estimate that, relative to the current baseline, a move to pristine standards at all sites would generate average recreational benefits of £17.89 per person per annum across the affected area. If we add in the value of avoiding the losses caused by climate change this value rises to £28.33.

Clearly both avoiding losses and generating overall improvements in water quality deliver substantial benefits. However, such policies would not be costless. Aside from any direct costs of water quality improvement, avoiding the land-use change induced by climate change would mean that farmers would forgo profits associated with moving to higher value arable production. To assess this we need to analyse the aggregate benefits to recreationalists from water quality improvements and compare these to the overall costs of compensating farmers for profits forgone.

Aggregation: Benefit-cost analysis of policy action

The recreation value model allows us to calculate aggregate values associated with both the climate change and WFD policy driver. Importantly, the model allows us to incorporate the very obvious distance decay in values illustrated in Figure 7. This is vital in order to avoid bias in aggregate value estimates and allow for both declines in value over distance and the heterogeneous distribution of populations across space (Bateman et al., 2006a).

Aggregation shows that the recreational value losses induced by climate change are concentrated in the west of the catchment in those areas which experience the largest decline in water quality, as shown in the upper row of Figure 7. Even though the local population density is considerably lower than in the east, this still generates an aggregate loss of approximately £26million per annum. Switching to consider the recreational benefits which are generated by implementing a WFD policy and improving water quality at all sites to pristine, the lower row of Figure 7 shows that these gains are concentrated in the eastern part of the catchment. The high population density in this area contributes to an aggregate benefit value of approximately £65million p.a. This in turn suggests that the combined value of avoiding climate change impacts and implementing WFD improvements would be in the order of £91million p.a.

In order to assess whether such a policy intervention would improve social wellbeing we also need to recognise that, aside from any direct costs associated with improving water quality, preventing the climate change induced shifts in agricultural land-use which resulted in water quality reductions would involve farmers forgoing an improvement in their incomes. To assess these losses, and the compensation required to induce farmers to forgo these changes, we examine the structure of farming in the Aire Catchment at present. This is detailed in Table 3 which shows that the catchment is dominated by grazing systems although with a significant minority of arable production.

Table 3: Typology of agricultural production in the Aire Catchment (hectares and livestock head per 2km grid)[1]

	1969	1988	2004	Mean	s.d.	Min	Max
Cereals	87.8	94.6	76.4	83.0	77.4	0	347
Oilseed Rape	0.1	8.5	13.3	6.9	12.3	0	125
Root crops	10.1	9.5	7.5	9.1	18.7	0	187
Temp. grassland	41.1	28.8	22.6	29.3	28.7	0	349
Perm. grassland	116.7	115.6	112.7	113.0	97.0	0	400
Rough grazing	47.1	39.6	40.5	44.0	100.0	0	400
Other	22.8	26.6	45.7	37.8	45.6	0	400
Total ag. land	325.6	323.2	318.7	323.1	96.9	1.25	400
Non ag. Land	74.4	76.8	81.3	76.9	23.1	0	398
Ha per 2km grid	400.0	400.0	400.0	400.0	-	-	-
Characteristics of gazing systems in the Aire catchment:							
Dairy (head)	87.1	71.5	62.0	74.1	99.1	0	1128
Beef (head)	151.4	149.8	89.9	144.9	123.8	0	1221
Sheep (head)	472.2	784.1	323.8	693.6	899	0	11289

Notes: 1. Data is taken from the Agricultural Census held at the Edina database available at http://agcensus.edina.ac.uk/. This data is held at a resolution of 2km grid squares (400 hectares)

Recent estimates suggest regional farm gross margins (FGM; a measure of profit ignoring fixed costs) of £200/ha/yr for livestock and £500/ha/yr for arable production in mainly uplands areas such as the Aire catchment (based on Redman, 2016). The catchment comprises just over 100,000 hectares (Agbotui et al., 2014). Using the figures given in Table 3 suggests that of this, roughly 80% is under agriculture, of which over 50,000 hectares is grazed with the remainder mainly under crops. To ensure that we do not under-estimate the compensation required by farmers, we can assume that climate change results in farmers changing all land which is currently grazed into arable production with FGM rising by £300/ha on this land. This would result in an aggregate increase in farm income of roughly £11million p.a. This in turn provides a generous estimate of the compensation required by farmers for not changing from grassland to arable production in response to climate change (not only does it assume that all such areas would have been so converted, it ignores any increase in long term fixed costs which this move might have induced). Paying this compensation would remove the main land-use change driver of water quality reduction arising from climate change. Assuming this would maintain rivers at their current quality avoids the loss of £26million per annum in recreational value which water quality reductions put at risk; a net benefit of some £15million per annum.

Calculating the costs of improving water quality from the current baseline to the pristine quality envisaged under the WFD objective is more complex, as full appraisal would need to consider nutrient reduction policies both for agriculture and households. Arguably we would also want to include microbial pollutants (Hampson et al., 2010) although, to date, these have received less policy attention. Accepting, therefore, that the following is an incomplete analysis, we can use the same assumptions as above to estimate the costs to farmers of reducing arable production and associated high nutrient loading into the Aire. Again ensuring that we do not under-estimate compensation costs, if we assume that all arable production in the catchment is switched into low input grazing systems then this reduces FGM by about £6million p.a. If this was sufficient to raise current water quality to pristine levels then this would generate recreational benefits of approximately £65million p.a.; a net benefit of £59million p.a. Furthermore, a policy to both avoid a climate change induced reduction in water quality from present day levels and then improving that quality to pristine, would yield a net benefit of £74million p.a. While we would need to set this against the non-agricultural costs of attaining the WFD objective these would be relatively minor in comparison to such a major net benefit.

VI. Mechanism for transfer of value

Funding for the improvements outlined above could be obtained from general taxation as per the previous case study. Here targeted payments to private landowners, such as farmers, could be made under a water stewardship scheme. Alternatively funding could be obtained from private water companies through a mixture of regulation and incentives. This latter route is considered in greater detail in our subsequent business to business case study.

VII. Monitoring and verification

The general mode of monitoring and verification employed in the UK for similar land-use decisions is through occasional on-site inspection. This is usually undertaken by an expert who visits the site to undertake observations and tests which assess the actions taken by the farmers. A problem with this method is that it is relatively expensive and time consuming to implement and other methods exist which could be employed to reduce these expenses:

- Satellites could be used to verify land-use changes made by farmers. This method would be particularly effective when the changes in land-use are easily observable from satellite pictures e.g. the change from arable to grassland-use. Drone technology might also play a role here.
- The development of cheaper individual water monitoring mechanisms should be encouraged so that information about water quality is available to both farmers and monitoring authorities. Products have been developed to monitor water quality (e.g. Schäfer et al., 2008), but a longer term goal would be to develop cheaper digital monitoring equipment which could relay data in real-time to monitoring authorities to help prevent and treat pollution. Such data could also be used to take preventative measures to reduce the costs associated with short term pollution incidents. For example, water abstraction inlets could be closed if prior warnings of major pesticide incidents were received.

VIII. Effectiveness

Effectiveness of model

The analysis made use of a simplified climate scenario for illustrative purposes. Future analyses could make use of a wider array of climate change data, as well as including any non-linear effects and uncertainties in the modelling process for both climate and land-use change. It should also account for seasonal variation within the model. In particular levels of rainfall, which vary across the country and throughout the year, directly influence the concentration levels of chlorophyll-a in the water.

Effectiveness analyses are complicated by the timescale of environmental changes. Lags commonly occur between policy decisions, implementation, behavioural consequences and environmental response.

IX. Key Lessons learned

While acknowledging that the direct market impacts of climate change on agriculture may be positive within the UK context (although not in many other countries), a central contribution of this research has been to demonstrate that focusing solely on these direct impacts paints a highly incomplete picture of the net impact of climate change. In our case study of the UK's River Aire, several key points have been made:

- It is important to understand and model the various direct and indirect effects that climate change would have on a catchment area. In this case study, three important effects have been considered:
    - Climate change directly causes a decline in the ecological quality of a river because a temperature rise of 1°C causes an increase in water temperature, which in turn has negative impacts on the native flora and fauna of the river.
    - Climate change indirectly causes a decline in the ecological quality of a river through induced land-use change, which causes a shift towards crops raising diffuse pollution of waterways.
    - A decline in the ecological quality of the river Aire would cause a decline in the recreational value of the river.
- A policy to offset the negative indirect effects of agriculture was examined. This found that the benefits of such a scheme significantly outweighed its costs.
- Methods for funding such change include both direct state support and business to business arrangements, as discussed in greater detail below.

Business to business funding of natural capital improvements: Catchment level reverse auctions

I. Description of the problem

While the majority of PES schemes involve public sector funding, there is no reason why this need always be the case. Where latent 'natural capital markets' exist (i.e. situations where there is a buyer and seller of natural capital related goods), or where regulation can create such markets, then private companies have the potential to increase revenues, reduce costs and raise profits through the provision of enhanced ecosystem services. Such incentives are important to consider in a world where public finance constraints bind and where the private sector has significant greater resources than Governments can command.

Water quality in rivers is of significant interest to private water companies who spend large amounts of money treating water to conform to legal standards. Avoiding pollution at source by paying farmers to reduce their inputs of pesticides, fertilizers and sediments can potentially reduce this cost significantly. There are also public benefits to reducing pollution where such improvements generate biodiversity, recreation and tourism benefits, boost other economic activity (e.g. fisheries), or reduce costs such as the need to dredge downstream waterways and ports.

One example of a private sector natural capital market for water improvement is provided by South West Water (SWW), a private water company which supplies the south west region of England. SWW's Upstream Thinking Initiative (UTI) engages directly with farmers providing contributions towards capital investments which both benefit farm activities and reduce agricultural diffuse pollution. In the early years of the UTI the company adopted an advisor led scheme (ALS). Here advisors from the Westcountry Rivers Trust (WRT; www.wrt.org.uk) would visit farms to suggest capital works which could reduce the environmental impact of the farms. The main advantage of this approach is that the advisors have local knowledge about the farms and land concerned and can therefore suggest appropriate investments for improving water quality. However, this approach also has some disadvantages. The ALS approach is labour intensive and hence costly, relying on advisors to both identify which farms to approach and what investments should be made. Furthermore it is the ALS advisors who select which farms to target. This decision is based upon their judgement regarding the impact of a scheme but is made without information regarding the variation in costs across farms. This means that value for money has little role to play in determining the allocation of support across farms. This is a problem given that lower value for money reduces the ability of the company to maximise reductions in diffuse pollution given available budgets. Indeed this information asymmetry means that SWW do not know farmers' true costs of switching into less pollution modes of production. This gives farmers the opportunity to overstate their costs and thus gain a higher payment from SWW.

In order to address this problem WRT entered into partnership with LEEP who applied economic, game-theoretic techniques, to design a natural capital market to provide cost-effective environmental improvements (Day et al., 2013; Smith, 2016). Through an application in the River Fowey catchment in Cornwall, UK, this market moved the payment arrangement from one where the farmer effectively dictated the price for their participation in a pollution reduction scheme, to a situation where the water company elicited competitive tenders from farmers to supply improved water quality. The resultant Fowey River Improvement Auction market worked as follows:

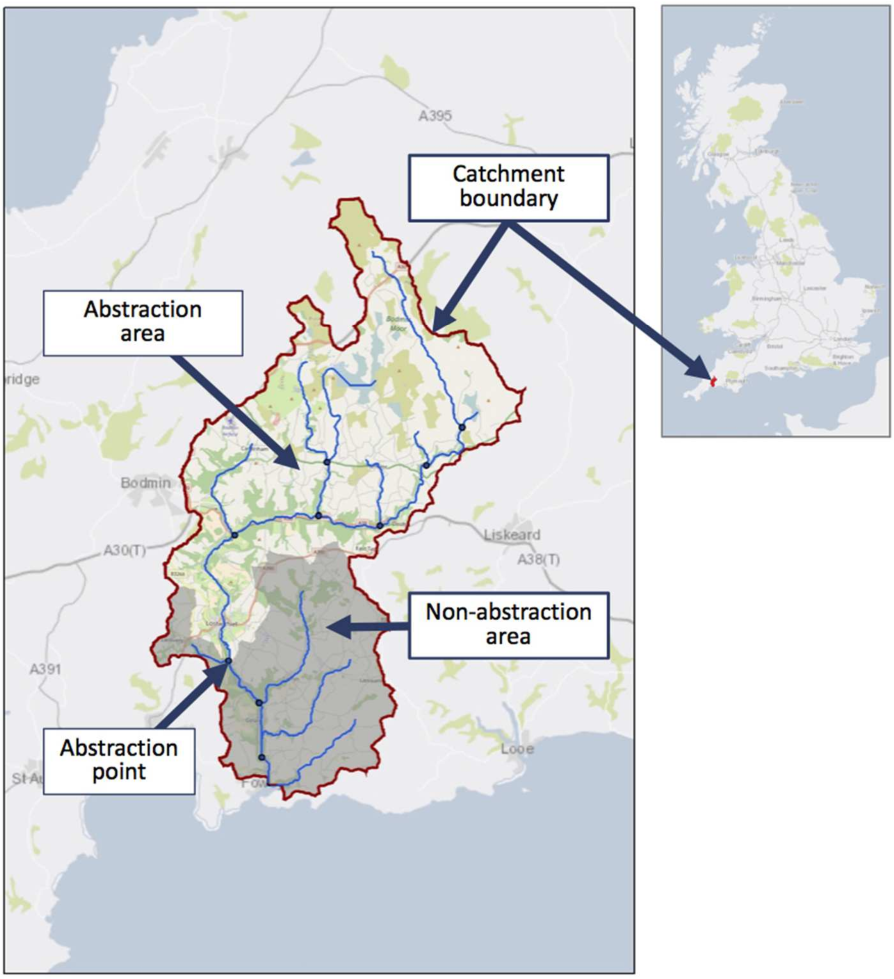
- Water quality researchers provided information regarding the best areas across the catchment for locating water improvement activities (e.g. on areas near to waterways or with certain vulnerabilities to pollution). This in effect provided a measure of investment benefits and was kept confidential to the water company and analysts;
- Farmers were then invited to prepare bids for investments from the water company for improvements to water related infrastructure on their farms (e.g. better hard standing, drainage, slurry stores, etc.);
- All farmers in the catchment were approached, thereby setting up competition for investments across farmers and driving down stated costs;
- All bids were compared with their expected benefits to allow the water company to derive a value for money assessment of each potential investment;
- Those bids delivering the highest value for money were funded.

This process, in which those bids offering the lowest cost per unit of benefit are funded, is commonly referred to as a 'reverse auction' (although strictly speaking such environmental scoring approaches are applicable to a variety of formats). To increase participation and further drive down costs, the market allowed a number of 'information rounds' prior to fixing final bids. Here, individual farmers were informed about whether, given other bids received, their bids had attained the levels of value for money likely to result in payments being made. This allowed farms to revise their bids either by reducing costs or altering investments to improve benefits.

II. Biophysical explanation of the relevant ecosystem services

Figure 8 shows the River Fowey and its abstraction area which supplies 97% of Cornwall's drinking water. The Fowey River Improvement Auction natural capital market applied to all farms within the catchment area upstream of the abstraction point.

Figure 8: The Fowey Catchment Area.



III. Identification of the beneficiaries

Clearly the water company can benefit from the reverse auction scheme. Reducing in-stream pollution loadings both reduced water treatment costs and generated a greater return on investment for SWW when compared with the ALS approach (SWW, 2017). This can generate a mix of increased profitability for the water company and/or lower bills for SWW customers.

Farmers' participation in the natural capital market shows that they also benefit from the scheme. While an individual who previously received payments under the ALS system may receive lower returns under the competitive market, overall this initiative has substantially increased the number of farmers engaged with SWW suggesting a greater degree of expected gains from the scheme.

The Fowey reverse auction also generates positive spill-over effects on the environment. Lower pollution loading in the river increases levels of dissolved oxygen and reduces eutrophication. Consequently, native flora and fauna, such as waterlilies and fish thrive, giving greater diversity to the ecosystem.

Improvements in the natural environment in turn generates the potential for gains in a range of ecosystem service related values including:

- Recreation activities, including in-river activities such as angling, boating and swimming, as well as informal recreation including walking, picnicking, etc.

- Indirect enhancements of economic activity including gains to the tourism sector and associated employment and businesses (pubs, hotels, markets) etc.

- Direct economic gains from improved water quality such as enhancements to commercial fishery and shellfishery operations. This can potentially be extended to lowering costs of dredging (clearing the river bed of mud and weeds) in the navigable river and ports if sediment deposition is reduced.

- Gains in non-use values associated with improvements in the provision and quality of wild species habitats and associated biodiversity.

IV. Identification of the suppliers

Any landowners and/or farmers who operate within the abstraction zone of the catchment have the potential to reduce diffuse pollution and hence supply cleaner water to SWW. Clearly this potential is mitigated by location within the abstraction area with those nearer to waterways being of greater importance in providing clean water.

V. *Quid pro quo*

South West Water distributed grants to farmers through the Fowey River Improvement Auction (Day et al., 2013) which used the reverse auction market to allocate funds to farmers for capital investments. The scheme sought to test whether improvements could be made relative to the ALS approach previously used to allocate funds. In order to design an efficient auction the analysts determined the following:

1. What the change was: SWW funding was provided for capital and operational investments on farmland, e.g. to update slurry storage, watercourse fencing, covered livestock feeding areas, management packages to test soil for runoff potential, waste management, and excluding livestock from watercourses.
2. How it would impact the water quality: These different capital investments would have different impacts on reducing pollution levels. The impact of the change depends in part upon the current state of the area which was being altered, e.g. further updating slurry storage on a farm which had recently replaced its slurry storage might represent poor value for money.
3. Where the change was happening: Investments in alternative locations generate different impacts on water quality, e.g. generally those near to the river have greater effect. It also mattered how pollution sensitive the area of change was, given that some land areas are more prone to pollution runoff than others. Factors affecting local sensitivity of an area include soil characteristics and gradient.
4. The likely costs of change – Includes the amount of capital investment required to implement the change by both the farmer and SWW and the cost of monitoring that the change has taken place.

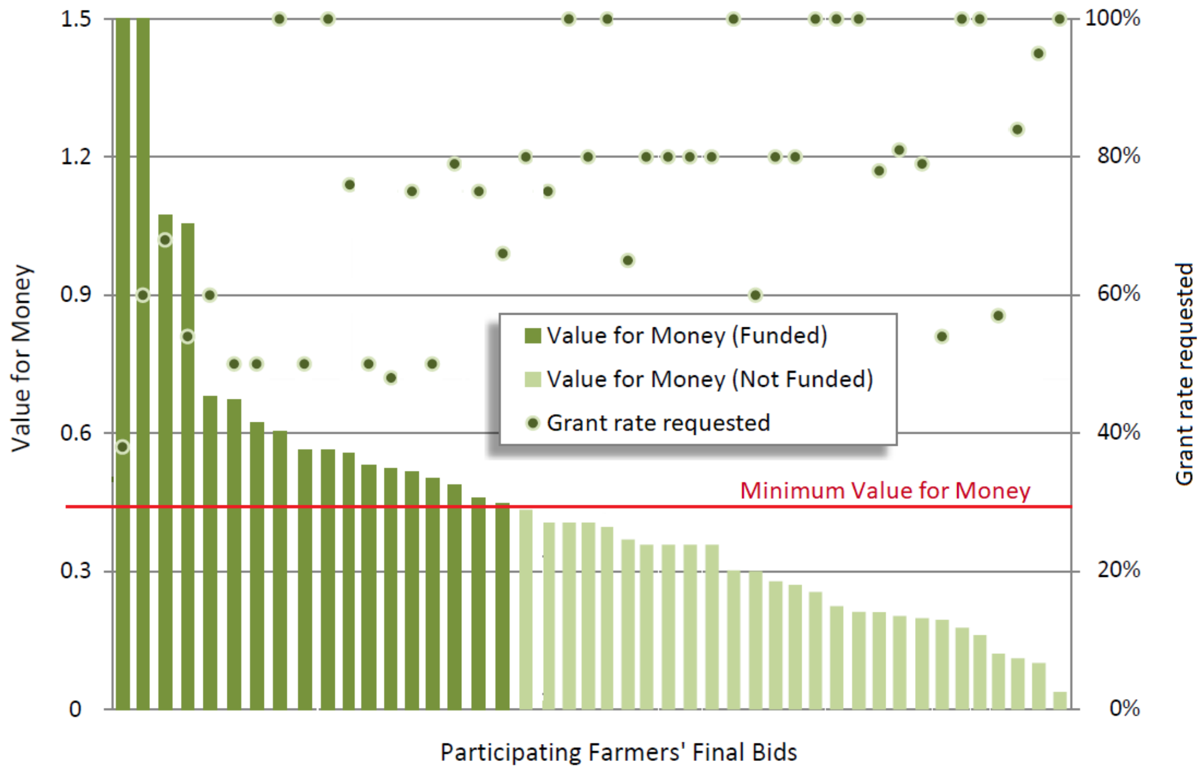
Taking this information together allowed the analysts to estimate a value for money score to each proposed investment. While the information regarding points 1-3 above could be obtained in the ALS, but point 4 could not. This represents a market failure in the ALS because the farmers were in a position where they do not have to reveal their real costs and indeed, can overstate them. This information asymmetry is likely to result in overpayments to individual farms which in turn reduces overall impact as less farms can be engaged for a given budget. This in turn further inhibits additional investment by the water company who receive comparative poor value for money from their investment.

The reverse auction provides farmers with an incentive to reveal their true costs. Given this information the water company was prepared to fund up to 100% of the cost of capital works. The rules of the auction were that farmers were informed that bids would be judged on a value for money basis, i.e. the likely benefits of changed behaviour were compared with the costs as offered by the farmer. By inviting multiple farmers to submit bids the buyer sets up a competitive market where farmers now have an incentive not to overstate their compensation requirements. A common feature of reverse auctions, as used in Fowey, is to allow farmers practice rounds where they submit bids and are told whether they are likely to be awarded the contract. Evidence has shown that reverse auction markets such as this deliver improved value for money for payees. This efficiency in turn allows payees to engage in further contracts, delivering far greater environmental improvements from the same budget.

Figure 9 shows the farmers' final bids ordered from left to right by value for money. The minimum value for money line indicates the point at which the available budget was exhausted. As can be seen, the competitive structure of the market ensures that all funded bids offer high value for money. A higher value for money score indicates better value for money (the value of 1 has no special significance and, in particular, it is not related to a 1:1 benefit cost ratio). Full details of the value for money scoring system are given in Day et al., (2013).

Figure 9 also shows the somewhat noisy relationship between value for money and the grant level requested. While a number of low value bids are ones which requests high levels of funding, and it is clear that some farms asking for lower grants (i.e. lower costs) deliver high value for money, nevertheless we can see that at almost any grant (cost) level there are some farms offering good value for money.

Figure 9: Comparison of Participating Farmers' Final Bids: Value for money and grant rate requested.



VI. Mechanism for transfer of value

Cash payments were made directly from SWW to funded farmers so that the capital investments could proceed promptly. These payments were made on the condition that the farmers signed and fulfilled a contract whose conditions and duration were determined by the nature of the capital works. Contracts typically lasted 25 years for more substantial, longer-lived investments (e.g. slurry storage and roofing) and 10 years for shorter-lived capital items (e.g. fencing and concreting). The contracts stipulated that the capital items in question would have to be used for the agreed purpose, be properly maintained and insured against any damage.

VII. Monitoring and verification

All of the capital investments were easy to monitor and verify because payments were issued on the actions taken by farmers. Visible confirmation of a build or management changes was all that was needed to verify actions had taken place.

An improvement might be made in the way water quality enhancements were verified in the reverse auction scenario by shifting from payments for action to payments for outcomes. The problem with payments for action is that there is no guarantee the water quality in the river will increase from the changes to capital made on the farms. Without measuring the outcome of the changes, it is also impossible to know how much effect each capital investment had. In principle, monitoring water quality in the catchment or at each farm in real time would determine the effectiveness of capital investments. An additional benefit is that this might also decrease the costs associated with farm verification and monitoring visits. Moreover, real time monitoring of pollutant levels can help prevent any major decreases in water quality arising from major events, such as storms, by acting as an early warning system. While ongoing improvements in monitoring technology will help greatly here, in practice it is likely that a mixture of payments for action and outcomes will persist, in part to defray the risks to farmers inhibiting participation.

VIII. Effectiveness

The reverse auction case study discussed here was a relatively low budget, proof of concept exercise which proved very successful and provides positive prospects for similar schemes to be run in the future. SWW invested £360,000 of funds in the exercise which was considerably oversubscribed, with bids for £776,000 worth of investment being received. The reverse auction approach to investment decisions was found to be 20-40% more cost-effective than the advisor led approach.

SWW conducted their own effectiveness analysis of the Fowey scheme. The company estimate that the value of stopping pollution at the source, as opposed to cleaning polluted water provided a 65:1 benefit cost ratio (Everard, 2014). This level of return out-competes almost any other potential investment the Company could make. As a result of this SWW expanded their UTI investment more than 30 fold to a 2015-20 budget of £11million, taking in 11 catchments across Devon and Cornwall (Brockett, 2015; SWW, 2017).

Despite this success there are limitations of the scheme centred around the difficulty of assessing the impacts the capital investments had on water quality. First, it was hard to assess whether the investments made were ones which would deliver the best water quality improvements on the farm. All of the investments were reviewed by an advisor and deemed to be ones that would deliver environmental improvements, but they also noted that the farmers only identified 54% of the projects on their farm which could deliver such improvements. A drawback of the reverse auction is that the farmers may well not have the knowledge that farm advisors have with respect to potential environmental investments. However, it is worth noting that this was the first time that farmers had to make such decisions and we expect that the knowledge regarding such investments would increase if further auctions took place. A future modification could be to produce a hybrid scheme where an advisor visited the farms and scored the investments and this assessment could contribute to the value for money score used for bids. Alternatively, there is the opportunity to offer training courses to farmers to help close the knowledge gap.

Secondly, it was important to determine whether the capital investments made were ones which the farmers would have made without the grants. If the farmers were going to carry out these capital investments regardless of the grants, then clearly funding these investments is not an efficient use of resources from SWW's point of view. Surveys taken after the auction showed that 62% of farmers stated they "would have undertaken investments irrespective of whether they received funding from SWW". However, almost all respondents caveated this statement with responses such as: "only when, and if, alternative finance became available to fund it"; "not now and probably a number of years in the future"; and "to a lower standard than proposed in the bid". At the very least, these grants pushed forward the priority of capital investments which benefit the environment.

Future schemes could also make use of the potential benefit to getting multiple purchasers involved. Sharing the cost of the scheme reduces the burden and risk on any one investor, making it more likely that the scheme would continue into the future. Increased financing can occur, allowing for an expansion of the scheme which can increase the impact it has on the environment. It can also broaden the range of investment goals within the scheme, e.g. reducing sedimentation levels, which can have a greater positive impact on the environment and recreation (Smith and Day, 2018).

IX. Key Lessons learned

The Fowey reverse auction natural capital market proved a success and helped eliminate some of the problems associated with the ALS. Several key lessons have been learned by this case study, including:

- Business to business investments can produce win-win outcomes. Specifically, benefitting:
  - The water company: engaging more farmers; increasing the value for money of investments; increasing water quality.
  - Farmers: providing a novel and long term guaranteed income stream;
  - The environment: increased water quality; enhanced biodiversity.
  - Society: the variety of use and non-use benefits associated with water quality improvements; potential reductions in water bills.
  - Local economy: enhanced fisheries and tourism; some cost savings.
- The reverse auction can be used to encourage the providers of environmental goods to reveal their costs to the investors.
- Valuing bids on a value for money basis helps ensure the best investments are made.
- Reverse auctions can be cheaper and more time efficient than pure ALS approaches for large scale investments.

Conclusions

The chapter has provided an overview of the natural capital approach to decision making and the key roles of scientific and economic information in enhancing such decisions. We also show the flexibility of PES designs across different scales (national, catchment and sub-catchment) and both public and private sector involvement. Taking this information together shows the potential to refine existing decision making approaches to deliver economically efficient environmental enhancements.

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