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Review

Incentives, land use, and ecosystem services: Synthesizing complex linkages

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ABSTRACT

Incentive schemes are increasingly used to motivate the supply of ecosystem services from agro-ecosystems through changes in land use and management. Here, I synthesize the complex effects of incentives on ecosystem services through their influence on land use and management. Linkages between incentives and land use change, and between land use change and ecosystem services can be one-to-many, many-to-one, and many-to-many. Change in land use and management can affect multiple ecosystem services, with both co-benefits and trade-offs. Incentives can motivate multiple changes in land use and management and multiple incentives often interact with both synergies and tensions in their effect upon ecosystem services. These vary over both space and time, and can be non-linear. Depending on incentive design, changes in ecosystem service supply can also have a feedback effect on incentive prices. I suggest that continued quantitative development is required to further explore these linkages: in the influence of incentives on land use change; in the impact of land use change on ecosystem services, and; in ecosystem service supply feedbacks on incentive prices. Quantifying and understanding these linkages is essential to progress more comprehensive analyses of the impact of incentives on ecosystem services, and the design of incentives capable of realizing synergies and avoiding tensions.

1. Introduction

Services from agro-ecosystems include a range of provisioning (e.g., food, fresh water, and bioenergy), regulating (e.g., climate, erosion, and pests), supporting (e.g., biogeochemical cycling, biodiversity/habitat), and cultural (e.g., recreation and education) services (Power, 2010; Swinton et al., 2007). Agricultural land use has degraded the soil, water, and biological assets in agro-ecosystems to such an extent that the restoration of natural capital and rehabilitation of ecosystem services through changes in land use and management is now a global priority (Ehrlich et al., 2012; Foley et al., 2011; Millennium Ecosystem Assessment, 2005). A primary reason for this degradation is the failure of agricultural commodity markets to internalize environmental costs associated with land use and management decisions (Lant et al., 2008). New market-based policy instruments – particularly financial incentives such as payments for ecosystem services – have emerged to redress these market failures (Farley and Costanza, 2010). Whilst market-based incentives remain one of the great hopes for the restoration of ecosystem services (Daily et al., 2009; Pascual and Perrings, 2007), the potential for inefficiencies and negative outcomes has also been recognized (Frame, 2011; Kinzig et al., 2011).

In agro-ecosystems, incentives influence ecosystem services through motivating changes in land use and management (Fig. 1). This chain of influence is complex because

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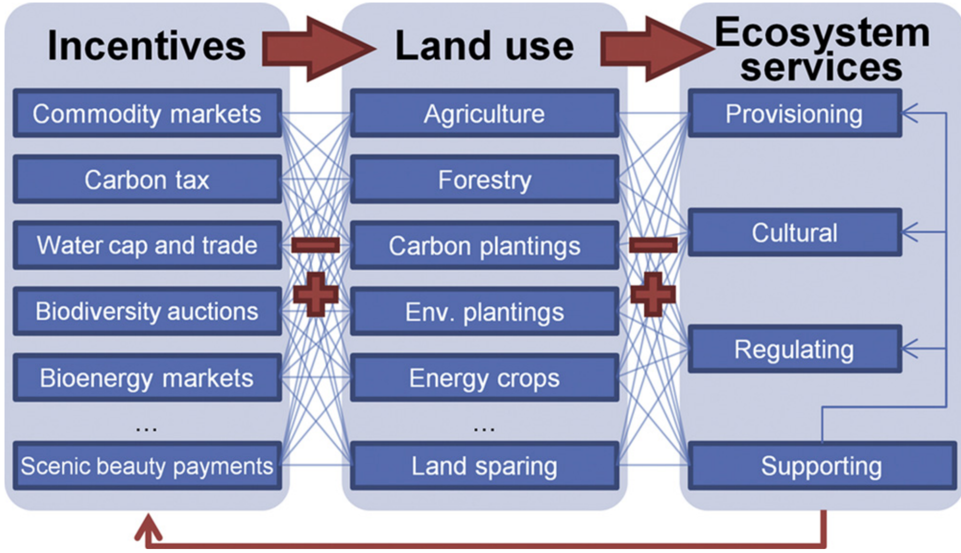

Fig. 1 – A simple conceptual representation of the linkages between incentives, land use, and ecosystem services. Financial incentives can have synergies (positive) and tensions (negative) in changing land use and management – which in turn have a range of co-benefits (positive) and trade-offs (negative) across multiple ecosystem services. Relationships between incentives and land use, and between land use and ecosystem services, vary across space and time and can be non-linear. These relationships can also be many-to-one, one-to-many, and many-to-many. The bottom link represents the potential dynamic effect of changes in supply of ecosystem services on incentive prices.

incentives can cause multiple intended and unintended changes in land use and management, each potentially having co-benefits and trade-offs across multiple ecosystem services (May and Spears, 2012). More often than not, multiple incentives co-exist (Pittock, 2011; Schrobback et al., 2011). These incentives interact, providing price signals for multiple land use and management changes, thereby compounding the effect on ecosystem services (Deal et al., 2012). Hence, the linkages between incentives and land use, and between land use and ecosystem services can be one-to-many, many-to-one, or many-to-many. These effects are typically heterogeneous across both space and time, and can be non-linear (Holland et al., 2011; Laterra et al., 2012). Changes in the supply of ecosystem services may also have a dynamic feedback effect on incentive prices, depending on instrument design. Understanding these effects can lead to substantial gains in the efficiency of policy and management in agro-ecosystems (White et al., 2012) and avoid negative outcomes (Bryan and Crossman, submitted for publication). Whilst many recent studies have addressed individual components, none have attempted the integrated assessment of incentive interactions on land use and ecosystem services inclusive of all of the linkages depicted in Fig. 1.

Here, I explore, clarify, and synthesize current understanding of the complex and multifarious influence of market-based incentives on land use and ecosystem services. I also discuss the requirements for quantifying these interactions and suggest directions for future work to support this important task. Awareness of these linkages is necessary to realize the benefits and avoid adverse outcomes for ecosystem services from changes in land use and management motivated by market-based incentives.

2. Incentives for ecosystem services

Ecosystem services contribute to human well-being through a range of direct-use (e.g., food and recreation), indirect-use (e.g., insurance and option), and non-use (e.g., existence, intrinsic and bequest) values (Pascual and Perrings, 2007). Whether or not the value of ecosystem services is reflected in markets depends on the *rivalness* of the good/service consumption (whether their use precludes use by others) and its *excludability* (whether access can be restricted to those who pay) (Kemkes et al., 2010). Some *market goods*, such as agricultural crops and livestock, are rival and excludable, and are routinely valued and traded in markets (Farley, 2008). *Public goods* (e.g., biodiversity), on the other hand, are non-rival and non-excludable; *common pool resources* (e.g., fisheries) are rival and non-excludable, and; *club goods* (e.g., toll access to a nature park) are non-rival and excludable (Kemkes et al., 2010). Markets for public goods and common pool resources rarely emerge naturally and, as farmers do not receive a price signal for these non-market ecosystem services, they under-produce them (Ribaudo et al., 2010).

Market-based incentives aim to correct this market failure and manage the supply of public good and common-pool type ecosystem services (Farley and Costanza, 2010). To be effective, incentives need to be supported by a carefully designed regulatory framework (e.g., safe minimum standards, quantifiable units of service provision, clearly defined property rights, monitoring requirements, and contractual arrangements) (Kroeger and Casey, 2007). Properly supported by regulation, financial incentives can be used to motivate the production of ecosystem services beyond critical levels by private individuals (Farley, 2008).

There are many different types of incentives for ecosystem services emerging at a range of scales (Farley and Costanza, 2010). These are commonly termed *payments for ecosystem services* or *agri-environment schemes*, and can be implemented through a range of instruments such as direct payments/rewards, tax incentives, cap and trade markets, voluntary markets, auctions, and certification programs (Pascual and Perrings, 2007; Pirard, 2012; Shelley, 2011; Yang et al., 2010). Globally, many such schemes have been implemented (Tallis et al., 2008). In any given region, multiple incentives – from global commodity markets to locally implemented incentives for public good and common pool resources – may co-exist for governing the production of ecosystem services.

3. Land use and ecosystem services

The type, intensity, and spatial arrangement of land use and management critically affects the type and amount of ecosystem services produced in agro-ecosystems (Goldstein et al., 2012; Raudsepp-Hearne et al., 2010). Changes in land use alter service provision (Dale and Polasky, 2007; Metzger et al., 2006; Nelson et al., 2010) either directly, or indirectly through effects on related services (Bennett et al., 2009). Multiple changes in land use and management can interact and influence individual ecosystem services. For example, changes in fertilizer application rates and stubble residue management in wheat cropping have been found to strongly affect soil carbon (Zhao et al., submitted for publication) (Fig. 2).

Similarly, individual changes in land use and management often affect multiple ecosystem services (Bennett et al., 2009; Bullock et al., 2011). Ecosystem service changes may be positively correlated such that changes in land use either increases or decreases their provision (Raudsepp-Hearne et al., 2010). Between these services, co-benefits can occur and the potential for win–win outcomes is greatest, but lose-lose outcomes are also possible (Tallis et al., 2008). However, negative correlations in ecosystem service changes may also occur (Raudsepp-Hearne et al., 2010). Trade-offs exist between these services whereby land use change can increase provision of one service, but only at the expense of others. Co-benefits and trade-offs can occur over multiple spatial and temporal scales (Power, 2010; Rodriguez et al., 2006) and vary over both space (Larsen et al., 2011) and time (Holland et al., 2011). Several win–win outcomes have been reported (Fig. 3a and b) (Dwyer et al., 2009), especially through the spatial targeting of hotspots of ecosystem service provision (Crossman and Bryan, 2009; Nelson et al., 2009). However, win–win opportunities are often hard to realize and trade-offs are the norm (Hirsch et al., 2011; Tallis et al., 2008).

A typical trade-off in agro-ecosystems is the replacement of many supporting, regulating, habitat, and cultural services provided by natural ecosystems for food, fiber, and increasingly, bio-energy services generated through agricultural production (Bennett and Balvanera, 2007; Lant et al., 2008). For example, specific trade-offs have been found between agricultural production and other ecosystem services such as

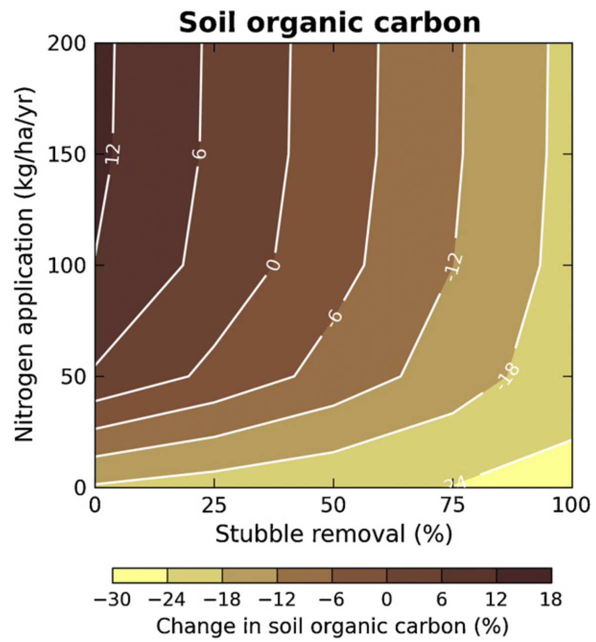

Fig. 2 – Complex influence of two land management changes in a wheat cropping system – stubble residue removal and nitrogen fertilizer application – on the soil organic carbon (SOC) storage ecosystem service in Australia's wheat-growing regions (Zhao et al., submitted for publication). This provides an example of the right hand side linkages in Fig. 1. Rates of change in SOC are median values across the region. SOC can maintained or increased through the application of ≥~40 kg/ha of nitrogen and ≤~35% removal of stubble residue. Decreasing nitrogen application rates ≤~50 kg/ha result in a sharp decrease in SOC at removal rates ≤~25%. Increasing nitrogen application rates does little to improve SOC at higher stubble removal rates.

sediment regulation (Swallow et al., 2009) and native species persistence (Barraquand and Martinet, 2011). When changing agricultural land use back to natural ecosystems through restoration, trade-offs have been found between achieving salinity and biodiversity objectives (Maron and Cockfield, 2008), and between carbon sequestration and a range of other services including biodiversity (Crossman et al., 2011b; Nelson et al., 2008), food (Nelson et al., 2010; Paterson and Bryan, 2012), and water (Chisholm, 2010) objectives (Fig. 3d). When multiple ecosystem services are considered, more efficient outcomes can be achieved where the net gains of land use change are maximized (Chen et al., 2010; Crossman and Bryan, 2009; Nelson et al., 2008; Wainger et al., 2010).

4. Incentives, land use change, and ecosystem services

Incentives are commonly designed to address a single ecosystem service following the Tinbergen principle (Tinbergen, 1952). The rationale is that individual policy instruments can rarely achieve multiple policy objectives efficiently (e.g.,

Fig. 3 – Australian examples of ecosystem services incentives with co-benefits and trade-offs. (a) Rice harvesting in the Murrumbidgee Irrigation Area, near Griffith, New South Wales. Rice and other crops such as cotton are opportunistic crops grown in wet seasons in Australia. By using surplus water to produce food and fiber services the trade-offs of reduced environmental flows for riparian ecosystems are minimized (photograph courtesy Willem van Aken). (b) Fire is a regular occurrence in the savannah ecosystems of northern Australia with implications for several ecosystem services dependent upon the season and hence, intensity of fire. Prescribed burning is actively used to reintroduce low intensity fire to the landscape which can enhance conservation values, protect property, improve cattle pasture, reduce carbon emissions, and involve Aboriginal traditional owners (photograph courtesy CSIRO Ecosystem Sciences). (c) Barley crop growing near Adelaide, South Australia. Natural ecosystems and the services they provide have been largely replaced by cereal cropping and livestock grazing for the production of food and fiber in the agricultural regions of southern Australia (photograph courtesy Christine Painter). (d) An ecological restoration project in bushland, Keilor, Victoria. With appropriate institutional rules in place, a carbon market could encourage large areas of ecological restoration with co-benefits for both carbon sequestration and biodiversity. Other benefits may include reduced erosion and sedimentation. However, caution is required to avoid impacts on food and fresh water services (photograph courtesy Nick Pitsas). (e) Tasmanian blue gum (*Eucalyptus globulus*) plantations growing on farmland in south-western Australia (8 years-old). Around 1 million hectares have been planted across southern Australia since 1998 following taxation incentives provided under the Managed Investment Act. Plantations provide carbon sequestration, timber, and some biodiversity benefits but reduce environmental flows and fresh water for human needs, and preclude agricultural production on the same land (photograph courtesy T. Grove). (f) A stockman musters cattle on Belmont Station in central Queensland. Beef cattle are one of the main agricultural industries across northern Australia, an industry strategically placed to service expanding markets in Asia. In response to emerging carbon market, the trade-off – significant greenhouse gas (methane) emissions from cattle, are being minimized through herd management and diet which is a focus of ongoing research (photograph courtesy CSIRO Livestock Industries).

Nelson et al., 2008). However, such interventions often have unanticipated consequences beyond their primary objective (Merton, 1936) (Fig. 4). The complexity of linkages makes this especially acute in the context of incentives for ecosystem services. As trade-offs between services in agro-ecosystems are common over space and time (Rodriguez et al., 2006), the failure to consider broad impacts in the design of incentives often leads to outcomes that are to the detriment of society. Unintended negative consequences of incentives for ecosystem services have been reported many times (Fig. 3e). For example, subsidies for motivating afforestation of agricultural land have actually been found to reduce carbon sequestration through shortening economically optimal rotation times (Tassone et al., 2004). Gren et al. (2010) found that whilst a single-objective payment for biodiversity achieved near the maximum possible social benefit over all services – when compensation was paid for producing a single non-market service (scenic beauty), the net social value produced across multiple services decreased. Biofuels markets have been found to generate carbon and energy benefits at the expense of food and fiber production (Bryan et al., 2010a).

Incentive design is increasingly seeking more efficient outcomes through bundling payments for multiple ecosystem services (Deal et al., 2012; Raudsepp-Hearne et al., 2010). Crossman et al. (2011a) designed a benefits metric for a conservation auction that included 23 landscape-scale and 14 site-scale indicators of natural capital. Wainger et al. (2010) targeted invasive species management payments for efficient production of habitat, property protection, forage, and hunting services. Wunscher et al. (2008) demonstrated substantial efficiency gains from targeting auction payments for scenic beauty, biodiversity, and water services in Costa Rica. Usually, services for which markets are more difficult to create (e.g., public goods such as biodiversity) are bundled with other more easily marketed services (e.g., carbon and recreation) (Wendland et al., 2010). Bundling can create price premiums for sellers but may increase transactions costs associated with monitoring spatially varying services (Kemkes et al., 2010). The bundling and supply of multiple ecosystem services has been accepted as a general principle for ecosystem service markets (Farley and Costanza, 2010).

Commonly, multiple incentives interact – and this affects changes in land use and management with flow-on impacts for ecosystem service provision (Fig. 5). Incentives for ecosystem services such as carbon, water, biodiversity, soil health, and bioenergy combine with established markets for agricultural commodities. Each incentive provides a price signal for changes in land use and management by landholders who make decisions in response to the totality of economic opportunities and risks (Fig. 3f). Landholders can take advantage of multiple markets for ecosystem services by *credit stacking* – the sale of the ecosystem services co-benefits generated through a single change in land use and management into multiple ecosystem services markets (Deal et al., 2012). However, incentive interactions are much more complex than this. Often unconsidered in credit stacking are the costs associated with ecosystem services trade-offs. Thus, to use the example of Deal et al. (2012), whilst a landholder restoring a hectare of riparian forest may be able to produce, stack, and sell credits simultaneously into water quality, carbon, and biodiversity markets, costs may also accrue to account for adversely impacted services such as a decrease in run-off.

Thus, whilst some incentives pull together toward achieving a policy objective synergistically, other instruments pull against each other, creating tensions. Several examples of tensions between incentives have been reported. The US federal Conservation Reserve Program paid people to retire environmentally sensitive land from agriculture whilst other federal farm subsidies sought to encourage agriculture. By raising the profitability of cropland through subsidies, the government directly competed with itself in providing incentives for landowners to retire land (Lubowski et al., 2008). A carbon price incentive was found to be less effective in motivating land use change when the costs of water used by reforested areas was accounted for (Chisholm, 2010). In the US, federal flood control and drainage programs provided opportunities for large scale conversion of wetlands to agriculture, working against wetland protection policies under the Clean Water Act (Stavins and Jaffe, 1990). Bryan and Kandulu (2011) documented a tension between taxation incentives that encourage landholders to increase cattle stocking densities and natural resource management payments aimed at sustainable land management.

In an integrated assessment of multiple incentives, land uses, and ecosystem services, Bryan and Crossman (submitted for publication) found complex interaction effects. Incentives for agricultural commodities, carbon, water, and biodiversity displayed synergies and tensions in their effect on food and fiber, carbon, water, and habitat services through their influence on agriculture and reforestation land uses (Fig. 5). The effects of incentive interactions across multiple ecosystem services were found to include non-linearities, interdependencies, and threshold effects (Bryan and Crossman, submitted for publication) (Fig. 6). These impacts of incentive interactions across multiple ecosystem services suggests that full knowledge of these consequences is essential to efficiently realize opportunities for synergies and minimize tensions. Bryan and Crossman (submitted for publication) however, did not assess feedbacks from changes in ecosystem service supply on incentive pricing (Fig. 1).

5. Directions for quantifying the influence of incentives on ecosystem services

To adequately quantify the complex influence of incentives on ecosystem services via land use change, the linkages in Fig. 1 need to be specifically addressed. We need to address the challenges of quantifying the influence of incentives on land use change, quantifying the impact of land use change on ecosystem services, and quantifying feedbacks from changes in ecosystem service supply on incentive prices. Below I discuss progress against these challenges and suggest priorities for future research.

Modeling the impact of incentives on land use change is based on the premise that regional patterns of land use change in agro-ecosystems emerge from micro-level land-use decisions by individual landholders. Incentives, with appropriate institutional support (e.g., regulatory frameworks and

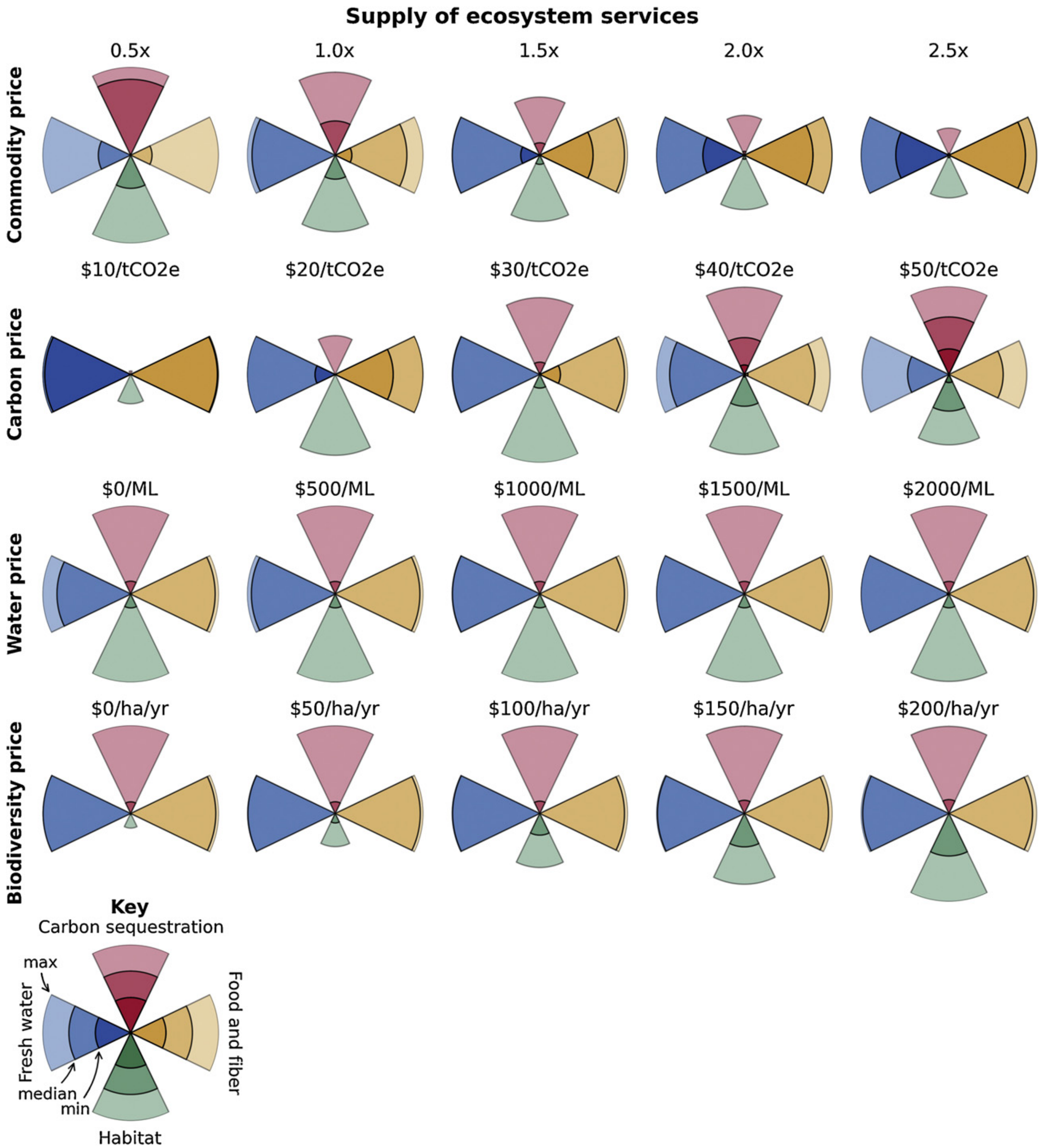

Fig. 4 – A synthesis of the production of multiple ecosystem services following land use change under various incentive prices for agricultural commodities, carbon, water, and biodiversity in the 15 million hectare agricultural region of South Australia as simulated under the median cost scenario of Bryan and Crossman (submitted for publication). The larger the fan blades and the deeper their color, the more of the ecosystem service provided. The fan blade lengths are relative, being linearly rescaled between the overall maximum and minimum values across 1875 scenarios, and include the minimum, median, and maximum scores representing the variation in ecosystem service provision. In this case study, markets affect the profitability and economic viability of three land uses – agriculture (wheat/sheep), carbon plantings, and environmental plantings. The spatial distribution of the most profitable land uses changes under different incentive price levels for agricultural commodities, carbon, water, and biodiversity – and this affects the provision of ecosystem services. An increase in agricultural commodity prices decreases the amount of carbon sequestered and habitat restored whilst increasing food production and fresh water provisioning. An increase in carbon price has the opposite effect. Increasing water price has a weak influence on increasing provision of fresh water with negligible influence on the other services. Likewise, increasing biodiversity price increases the provision of habitat services exclusively. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

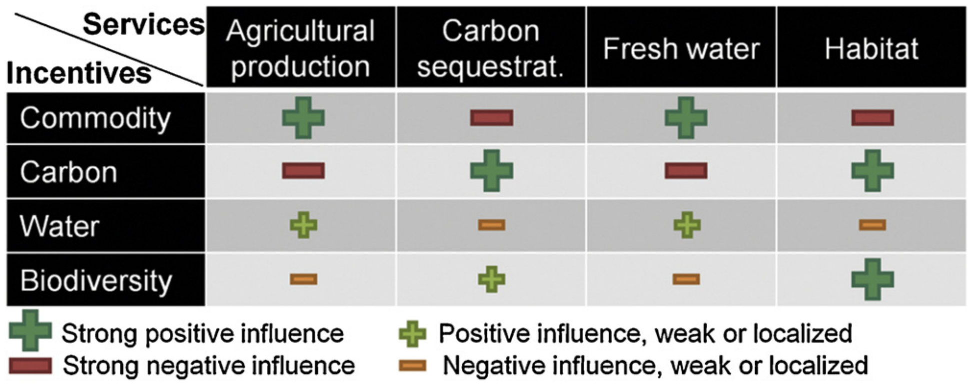

Fig. 5 – Interactions between incentives and the effects on ecosystem services in the South Australian agricultural regions summarized from Bryan and Crossman (submitted for publication).

information), change the relative profitability of land uses and provide a price signal for landholders to change land use (Irwin and Geoghegan, 2001; Lewis et al., 2011; Lubowski et al., 2008). Carbon markets, in particular, can provide economic opportunities for landholders to convert agricultural land to tree-based land uses (Alig et al., 2010; Bryan et al., 2008; Harper et al., 2007). Alig et al. (2010) found that carbon-related payments to landowners can have substantial impacts on future patterns of forestry and agricultural land use, levels of terrestrial carbon sequestration, forest resource conditions, agricultural production trends, and bioenergy production. Conversely, reductions in agricultural prices can also lead to the retirement of agricultural land and conversion to forest (Vuichard et al., 2008). Profitability has been widely used to evaluate the competitiveness of alternative land uses (Hunt, 2008; Maraseni and Cockfield, 2011; Wise and Cacho, 2011) and to quantify the impact of incentives on ecosystem services (Antle and Stoorvogel, 2006; Bryan et al., 2010b, 2008; Dymond et al., 2012; Polasky et al., 2008; Townsend et al., 2012). Variation in economic parameters including discount rates, upfront establishment costs, and ongoing transactions and maintenance costs are well known to affect economic returns from land use (Bryan et al., 2008).

However, whilst profitability is known to be a major driver of land use change and adoption of conservation technologies (Lubowski et al., 2008), a range of other less-well-known factors are also important. Uncertainty, risk, and option values are important given the uncertainty and irreversibility of investment in land use change, constraints on labor and capital, and a range of other unmodeled costs and benefits all affect the magnitude and rate of potential land use change (Lubowski et al., 2008; Stavins, 1999). These factors can result in less land use change than would be predicted by economic theory and cause the undersupply of ecosystem services. However, whilst most landholders may be unwilling to reforest their land without financial incentives – for some, the level of incentive required could be less than the net returns to current agricultural activities on marginal agricultural land (Shaikh et al., 2007), as they receive benefits from growing trees that are not captured in market transactions. These benefits relate to potential reductions in risk from assured annual payments, and the provision of ecosystem services particularly those that may help sustain agricultural production, but also aesthetic benefits, bequest value, and other benefits.

The many determinants of land use change make predicting land use change decisions in response to incentives extremely challenging. Previous studies have used a revealed preference econometric approach based on data from

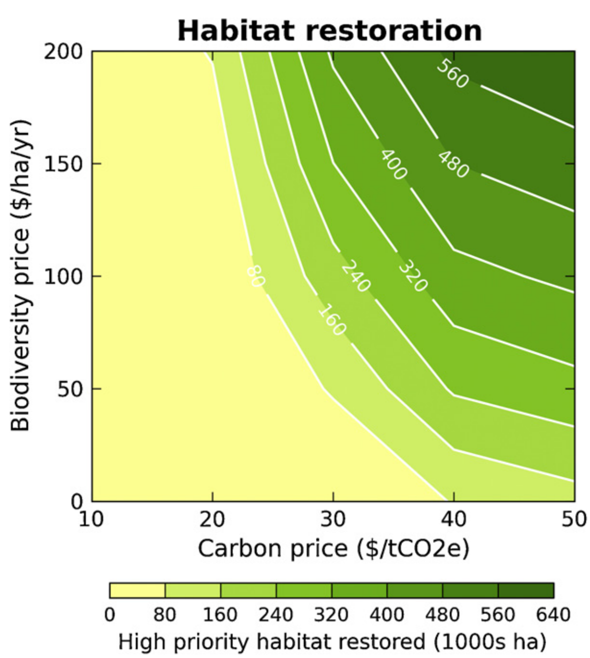

Fig. 6 – Interaction between a biodiversity payment and a carbon price incentive and the effect on the supply of habitat services, through motivating widespread adoption of environmental plantings in the 15 million hectare agricultural region of South Australia. At a low carbon price ($\leq$20\$/t$CO_2{}^{-e}$), very high biodiversity payments were required to induce adoption of environmental plantings in high priority areas. At higher carbon prices, a biodiversity payment is an economically effective tool for motivating adoption.

programs such as the Conservation Reserve Program to capture the effects of incentives on land use change and ecosystem services (Busch et al., 2012; Lewis et al., 2011; Lubowski et al., 2006; Nelson et al., 2008). The two great advantages of the revealed preference approach lie in its empirical basis and its inherent ability to capture the influence of multiple drivers in projecting land use change. There are however, two outstanding questions about its application. Firstly, to what extent is the past able to predict the future? A complex systems view of land use suggests that non-linear effects such as surprises, dependencies, and threshold effects may limit the predictability of past responses and demand new approaches to land use change analyses (Briassoulis, 2008; Dawson et al., 2010; Parker et al., 2008; Parrott and Meyer, 2012). Secondly, for most regions, the data required to build revealed preference models do not exist. To compensate for this, minimum data methods have been developed. Compared to revealed preference approaches, minimum data methods can provide land use change models acceptable for use in policy analysis (Antle and Valdivia, 2006). New methods combining the strengths of complex systems and econometric approaches are urgently required to support future assessments of market impacts on land use and ecosystem services.

In modeling the impact of changes in land use and management on ecosystem services, substantial recent advances have been made and this is where the science underpinning the linkages in Fig. 1 is most developed. Several studies have measured, mapped, and modeled the spatial distribution of ecosystem services produced from various land uses and these efforts have recently increased in sophistication. For example, biophysical process models have been commonly used to map provisioning and regulating services such as food, bioenergy, water, and carbon (Crossman and Bryan, 2009; Crossman et al., 2011b; Paterson and Bryan, 2012; Stoms et al., 2012). Recent calls have been made for advances in the mapping of cultural services (Chan et al., 2012; Daniel et al., 2012). More sophisticated spatial properties such as supply, demand, flow, beneficiaries, and benefits transfer have been incorporated into the mapping of ecosystem services and quantifying their benefits and costs (Burkhard et al., 2012; Crossman et al., submitted for publication; Eigenbrod et al., 2010; Fisher et al., 2011; Syrbe and Walz, 2012). More focused progress in mapping the impact of land use change on ecosystem services is required for the accurate and meaningful assessment of the impact of market-based policy incentives.

To complement these advances, we also need to capture the dynamic feedback relationships between ecosystem service supply and incentive prices. This is where the science required to underpin the linkages in Fig. 1 is least developed. Prices for marketed services such as agricultural commodities are dynamic, responding to relative changes in supply and demand (Fig. 1). For example, given a constant demand, a decrease in the supply of wheat production through land use change such as the reforestation of agricultural land will place upward pressure on wheat prices (Wright, 2012). Incentives for non-marketed services can be also designed with flexible (or *elastic*) prices (e.g., as cap and trade instruments) that respond to changes in supply and demand of ecosystem services (Sterner, 2003). To apply the same economic principles to non-marketed services – in a water market for example, water prices should respond to the scarcity of the resource. Prices should increase in times of drought, thereby discouraging use and encouraging efficiency in irrigated agriculture, and ensuring flows for the continued production of other services from water-dependent ecosystems. Conversely, in wet periods, falling water prices signal increased availability of fresh water services for irrigation and other human uses. Such price elasticities in water markets have been reported in Australia's Goulburn–Murray district (Wheeler et al., 2008) and the US Rio Grande (De Mouche et al., 2011). Working efficiently, the theoretical outcome of these market forces will be a socially optimal supply of ecosystem services. However, this often fails in practice due to many factors including thin markets, differences in scale, levels of information, price volatility and risk, leakage, and institutional design (De Mouche et al., 2011; Sovacool, 2011). Incentive price dynamics, feedbacks, and market behavior needs to be quantified using partial and general equilibrium models (van der Werf and Peterson, 2009) for a complete understanding of the influence of incentives on land use and ecosystem services. This is a frontier of ecosystem services science.

6. Conclusion

Market-based incentives are widely used to govern the supply of ecosystem services from agro-ecosystems. In particular, incentives such as payments for ecosystem services are increasingly used to rebalance the supply of non-marketed services. The influence of incentives on ecosystem services is indirect, occurring through their ability to motivate changes in land use and management. I have shown that the relationships between incentives and land use, and between land use and ecosystem services are complex. These relationships may be one-to-many, many-to-one, or many-to-many. These relationships can be non-linear and vary across both space and time. Multiple incentives interact, with synergies and tensions in their influence on multiple ecosystem services. Depending on incentive design, there may also be a dynamic feedback on price through changes in the supply of ecosystem services. Whilst many studies have addressed individual components, no studies have attempted an integrated assessment of the interactions between multiple incentives, land uses, and ecosystem services. I discussed the need for continued advancement in three areas: quantifying the link between incentives and land use change decisions; quantifying the impact of land use change on ecosystem services, and; quantifying the dynamic incentive price feedbacks from changes in ecosystem service supply. These developments are badly needed to support the design of incentives that can capture the synergistic effects and avoid tensions between incentives in their influence on ecosystem services.

Conflict of interest

The author declares no conflict of interest.

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