Agriculture, Ecosystems and Environment 139 (2010) 1–12

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Review

Examining links between soil management, soil health, and public benefits in agricultural landscapes: An Australian perspective

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ARTICLE INFO

Article history:
Received 10 March 2010
Received in revised form 29 June 2010
Accepted 30 June 2010
Available online 23 July 2010

Keywords:
Ecosystem services
Ecosystem disservices
Public benefit
Soil health
Soil indicators
Soil management

ABSTRACT

Public expectations of soil management are gradually expanding beyond traditional primary production requirements to include diverse ecosystem services. In Australia, as in many other countries, the accommodation of these new expectations will require shifts in the practice of private land managers. In turn, this may require public intervention and the expenditure of public funds. However, public net benefits from soil management interventions are rarely established, in part due to a lack of understanding of the conceptual links between management changes, soil health, and associated services and benefits. This paper uses an ecosystem services-based approach to examine these links from an Australian perspective.

Entrenchment of the popular soil health concept in field-based assessments of agricultural production potential was found to limit the concept's applicability to questions of broader public benefit. Without expanding soil health to include more ecological indicators, the concept risks remaining peripheral to contemporary visions of multiple-outcome soil management in Australia. Conceptual and case study links were examined between soil properties and processes, soil-based services, and private and public net benefits. In this framework, benefits were produced from services, and were considered a more tangible point for public understanding and valuation than services. The qualitative case study highlighted many knowledge gaps relating to non-agricultural services and benefits from soils, particularly in the scaling-up of sub-paddock measurements, and in the form and constancy of relationships among services and benefits. Criteria for identifying priority public benefits from soil management were examined, namely, likelihood, degree, consequence, scale, direction, time lag, and valuation. Assumptions about these criteria require rigorous testing so that the what, where, when, and how of public benefits from changed soil management can be more clearly defined.

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doi:10.1016/j.agee.2010.06.017

1. Introduction

Soil is that 'invaluable, diverse, and fragile natural resource at Earth's terrestrial surface that provides for life support' (Wilding and Lin, 2006). Most appreciated for its role as a medium for providing nutrients and water to agricultural plants, soil is equally fundamental to a range of services including carbon sequestration, water quality and flow regulation, remediation of wastes and pollutants, and habitat provision for soil biota (Costanza et al., 1997; Daily et al., 1997; Lal, 2004).

While the importance of soil to life is indisputable, soil resources worldwide continue to degrade. The problem of ongoing soil degradation becomes particularly critical given projections of future global food requirements – e.g. 63% increase in average cereal yields by 2050 – most of which need to be met by land already under agriculture (Lal, 2009). This has resonance in Australia where expanding food markets in Asia present considerable export opportunities, but soil degradation remains a 'very significant' problem that is likely to intensify under climate change (Campbell, 2008).

International commitment to addressing soil degradation and improving soil management is evident in various government programs and strategies. One enduring example is the United State's Conservation Reserve Program, which pays private landholders to retire erosion-prone soils from crop production, with the demonstrated aim of improving the joint production of soil conservation, farm income, and water quality in agricultural landscapes (Lant et al., 2005). At a broader policy level, the European Commission (EC) has adopted a 'Thematic Strategy for Soil Protection' (Commission of the European Communities, 2006), which has led to the development of national policy statements like England's recent 'Safeguarding our Soils' strategy (Defra, 2009). However, soils are not receiving such strong policy interest in all parts of the world. In Australia, for example, government focus on soil conservation has decreased in the last two decades (Campbell, 2008), despite substantial increases over the same period in federal government expenditure on natural resource management programs (Hajkowicz, 2009).

It has been suggested that one factor that has contributed to the loss of focus on soil conservation in Australia is an apparent failure to 'join the dots' between good soil management and broader environmental, societal, and economic outcomes (Campbell, 2008). This indicates a lack of clarity on links between paddock-level aspirations for soil management – often represented by the soil health concept (MacEwan, 2007; Kibblewhite et al., 2008) – and broader expectations, like those encapsulated in the concepts of ecosystem services and human welfare benefits (Millennium Ecosystem Assessment, 2005; Fisher et al., 2009). Thus, even at a conceptual level, it is difficult to answer the question 'what will we get for expending public (government) funds on soil management?' (Hajkowicz, 2009). This disconnection between on-site management and broader public benefits is a key impediment to defining realistic goals for soil conservation policy in Australia, and, as in natural resource programs worldwide, to clearly linking expenditure with tangible outcomes (Claassen et al., 2008; Hajkowicz, 2009).

Soil-based ecosystem services were implicitly acknowledged in the seven broad soil 'functions' of the EC's Thematic Strategy (Commission of the European Communities, 2006). This acknowledgement reflects growing recognition of the links between land degradation and global public good (Pagiola, 1999; FAO, 2002). However, it is only recently that broad links between the concepts of soil health and ecosystem services have been explicitly examined (Robinson et al., 2009). Moreover, while broad-scale costs and benefits of addressing soil degradation have previously been considered (FAO, 2001), few studies have integrated service-based frameworks into cost-benefit analyses of soil management.

This paper uses a service-based approach to examine links between soil management, soil health, and public benefits in Australian agricultural landscapes. First, it expands on the context of public intervention in (mostly private) soil management, and examines the place of the soil health concept within a service/benefits framework. Soil-based ecosystem services and disservices are then identified, and broad conceptual links with defined public benefits are established. These links are then applied to a regional case study that evaluates potential public benefits from soil management change. This regional-level approach is consistent with recommendations for implementing the EC's Thematic Strategy (Bouma and Droogers, 2007), with the clear difference that it highlights soil-derived benefits rather than soil threats, thereby supporting a shift away from a common damage-centric focus (Defra, 2007). The case study highlights key knowledge gaps in estimating both public and private net benefits from changed soil management, including the need for criteria to identify priority public benefits at policy-relevant scales. The paper aims to contribute to a new narrative on the importance of better soil management in Australia (Campbell, 2008), and to provide a stronger basis for articulating objectives and anticipated outcomes in public policies for soil conservation.

2. The context: public benefits from private soil management

It is inevitable that many of the Earth's soils will continue to be managed with a strong production focus. Agriculture remains the main land use in many countries (Hamblin, 2009), and has transformed about one-third of the Earth's land surface (Vitousek et al., 1997). Globally, the main agricultural practices of cropping and grazing account for 78% of human appropriation of net primary production (Haberl et al., 2007). Strong demand for food and fiber is set to increase given projections of a rapidly expanding human population (Matson et al., 1997). Production pressures on soils are certain against a backdrop of continuing low food prices, rising input costs, and ongoing pressures to exploit the soil capital in pursuit of short-term economic gain (Tilman et al., 2002).

In addition to production requirements, public expectations of natural resources like soils are expanding due to increasing awareness of 'ecosystem services' (e.g. carbon sequestration, water quality regulation, water yield), which provide the many benefits that humans derive from natural systems (Costanza et al., 1997; Burger, 2009). Just how agricultural landscapes should be managed to meet the dual challenges of production and ecosystem services is an issue of ongoing discussion. Some advocate retirement of non-productive agricultural land (Hamblin, 2009), and/or increasing yields from productive land to reduce the need to convert remaining native systems (Green et al., 2005). However, this 'land sparing' approach ignores probable increases in negative off-site effects associated with more concentrated inputs of water and nutrients (Matson and Vitousek, 2006), leading to arguments that agricultural land should be less intensively managed as part of a 'wildlife friendly' matrix (Vandermeer and Perfecto, 2007).

Whichever the land-use configuration, it is often the case that shifts in agricultural management to meet public expectations require shifts in the practice of private land managers. Unfortunately, there are very few circumstances under which private managers are able or willing to make substantial personal investment for the greater good (Lant et al., 2005), particularly where there are significant production opportunity costs (House et al., 2008). This realization has led to ongoing calls for publicly-funded instruments of change, often in the form of incentive payments for ecosystem services (Tilman et al., 2002; Harvey et al., 2008; Hamblin, 2009). Nonetheless, others warn that payments are not a silver bullet, and that a range of mechanisms for public intervention should be maintained (Redford and Adams, 2009).

Most if not all forms of public intervention in soil management – including payments and penalties, regulation, extension services, and research and development – require expenditure of public funds. In theory, but rarely in practice, this requires clear demonstration that public benefits outweigh costs (Craemer and Barber, 2007). Clearly, since public intervention often involves change in practice, associated change in net benefits (e.g. improved aesthetics) are of greater interest than measures of full benefit (full aesthetics), which are often impossible to quantify (Costanza et al., 1997). This information is key to choosing the most appropriate form of public intervention (Pannell, 2008), and seems critical for soils in current 'triage' approaches to natural resource investment (Bottrill et al., 2008), because the values of soils are largely hidden and are usually less appreciated than those of above-ground assets (Brussaard et al., 2007).

3. Soil health: can it be linked to public benefits?

'Soil health' is a term that has been widely used in the context of production agriculture (Kibblewhite et al., 2008). Here, the concept's relevance to a broader context is examined; that is, are there clear conceptual links between soil health, non-agricultural ecosystem services, and associated public benefits?

A frequently cited definition of soil health comes from Doran et al. (1996):

> '[soil health is] the continued capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, maintain the quality of air and water environments, and promote plant, animal, and human health'.

This represents an integration of biological with physical and chemical domains (Idowu et al., 2008), reflecting a recent emphasis on soil as a living system '... distinguished from weathered rock (regolith) mainly by its biology' (Kibblewhite et al., 2008).

The above definition of soil health was based on one of soil quality in the same paper (Doran et al., 1996). Both terms had their beginnings in field-based interpretations of a soil's capacity to produce agricultural goods (Doran et al., 1996; Carter et al., 1997). They are often used interchangeably (e.g. Idowu et al., 2008; Kibblewhite et al., 2008), although some suggest soil quality reflects capacity for an intended use as determined by parent material, topography, and climate (Doran et al., 1996; Carter et al., 1997), whereas soil health relates more to current condition or state, which reflects management effects within innate soil quality boundaries (Doran et al., 1996). These conceptual links between the two terms in their traditional agricultural context are summarized in the top portion of Fig. 1.

As the definition by Doran et al. (1996) highlights, soil health is context-dependent ('...within ecosystem and land-use boundaries...'). In agricultural terms, this often means that soil considered healthy in a particular bioregion for the purpose of one land use (e.g. dairy farming) might be considered less healthy for another (e.g. viticulture; Idowu et al., 2008). Thus, unlike measures of air and water quality, soil health cannot be defined in terms of a pure state (Letey et al., 2003). Rather, application of the traditional soil health concept requires different standards according to the many different combinations of land use and environment. In addition to agricultural production, recent definitions of soil health emphasize expectations that soils will simultaneously provide a range of other ecosystem services (e.g. water quality regulation, weather regulation, habitat provision; Kibblewhite et al., 2008). Thus, in addition to addressing land-use and environmental constraints, the contextual boundaries of soil health must also apparently accommodate societal goals for specific landscapes (Doran et al., 1996).

Despite broad aspirations for soil health, a recent survey of Australian landholders indicated continued reliance on a handful of mostly chemical indicators that have an established relationship with farm productivity (Kelly et al., 2009). This is consistent with this paper's tally of readily available soil information (in published papers or on soil agency websites) relevant to two contrasting soil scenarios in Victoria, south-eastern Australia. Here, 75–85% of data sources relevant to Calcarosol soils in the Murray Mallee Bioregion and to Ferrosol soils in the Strzelecki Ranges Bioregion (DNRE, 1997; Isbell, 2002) contained chemical and/or physical indicators, compared with 13–15% for biological indicators. Across the two scenarios, the most frequently reported individual indicators were, in decreasing order, pH, texture, organic carbon, water content, electrical conductivity, exchangeable cations, and bulk density. The predominance of physico-chemical indicators reflects recommended minimum data sets for land resource surveys in Australia (McKenzie and Ryan, 2008), and is consistent with indicator application frequency in worldwide environmental/soil monitoring programs (Winder, 2003). A comparative lack of biological indicators reflects lower accessibility and less standardization of biological methods, particularly emerging molecular-based methods (Bastida et al., 2008; Ritz et al., 2009).

A key selection criterion for soil indicators is their capacity to represent the service in question (Doran et al., 1996; Doran and Zeiss, 2000). Soil organic carbon, for example, is a 'star' indicator in agricultural soils (Bastida et al., 2008), largely due to its' frequently documented positive relationship with crop yields (Doran et al., 1996). However, empirically based links with crop response are usually lacking for most soil indicators (Carter et al., 1997). Moreover, any such links between crop yields and soil indicators are unlikely to be simple linear relationships that support a 'more is better' paradigm (Doran et al., 1996). High soil organic carbon, for example, can represent sub-optimal soil health in some situations where it leads to decreased available nitrogen for crop production relative to lower carbon systems (Letey et al., 2003), or increased application requirements of soil-incorporated pesticides (Sojka et al., 2003).

Despite attempts to formulate generalized relationships (Lilburne et al., 2004), the capability of common soil indicators to represent non-agricultural services remains largely unknown (Palm et al., 2007). Addressing this knowledge gap will require not only expanded evaluation of traditional indicators but also development and testing of 'ecological' soil indicators that more clearly represent properties and processes relevant to broader services. Here, several authors advocate a greater emphasis on biological indicators (Bastida et al., 2008; Mele and Crowley, 2008; Ritz et al., 2009) since 'the biota plays such fundamental roles in the majority of ecosystem services provided by soils' (Ritz et al., 2009). Others highlight an ongoing need for the development of more integrative indices (Kibblewhite et al., 2008) that involve various combinations of any number of sub-indicators (Bastida et al., 2008), including those chosen to reflect key soil-based ecosystem services (Velasquez et al., 2007).

Until there is greater capacity for ecological interpretation of soil indicators, traditional index-based assessments of soil health will remain peripheral to questions of broader public benefit. Conceptual links between change in soil management and associated benefits (Fig. 1) recognize that goals for sustainable soil management in an agricultural context are often expressed in terms of soil properties (e.g. to maintain or improve organic matter content). As indicated above, soil properties are typically represented by indicators of agricultural production potential (Carter et al.,

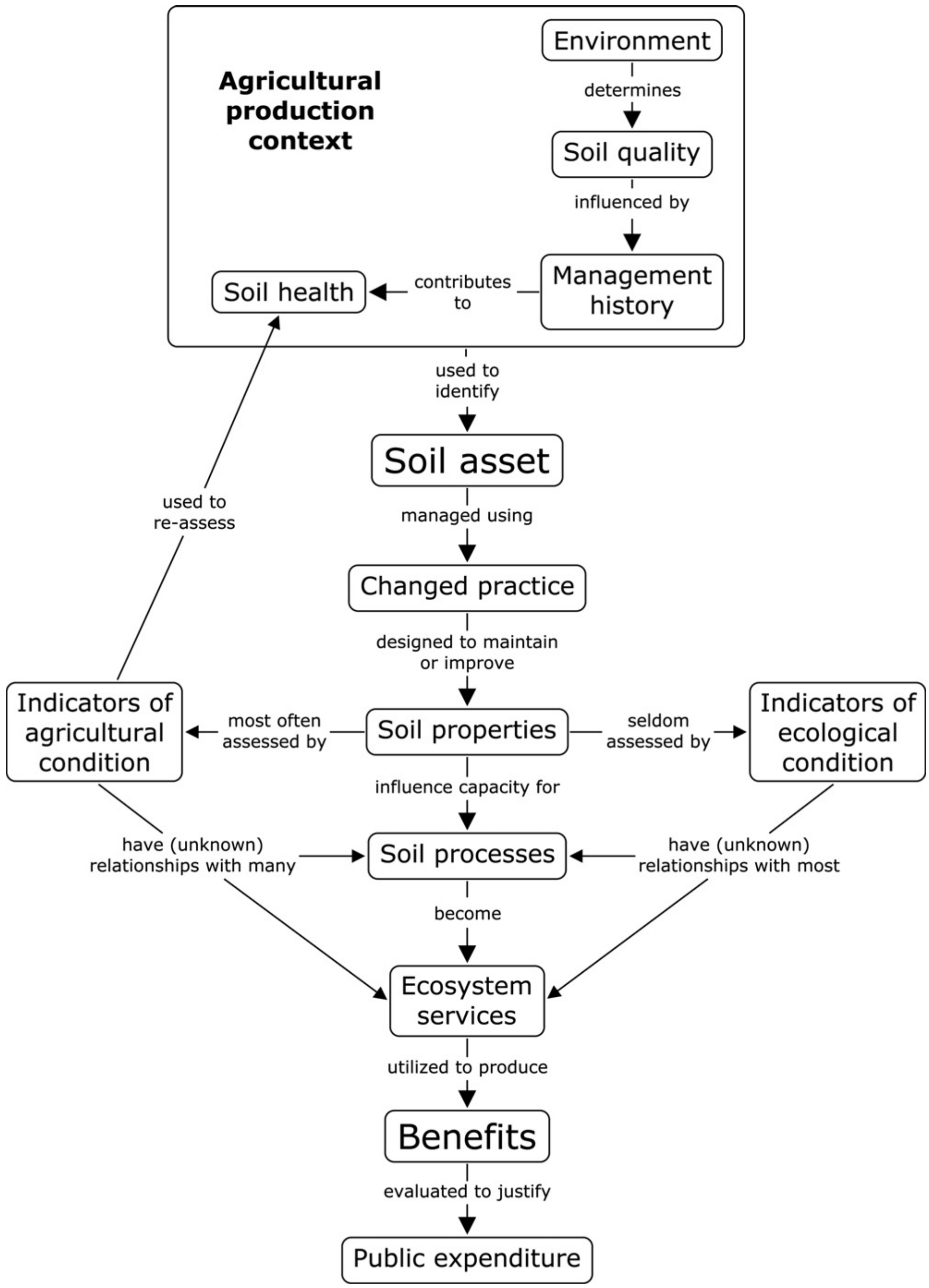

Fig. 1. Are there clear links between soil health and soil-based benefits to justify public expenditure on changed practice? Soil health is most often used to identify soil assets for agricultural production. New management practices are designed to maintain or improve soil properties, which influence the capacity for soil processes that become ecosystem services if they are utilized to produce benefits. Soil health is most often assessed using indicators of agricultural condition, which have largely unknown relationships with many if not most ecosystem services. This is also true for new 'ecological' indicators that might be used to expand the soil health concept. Thus, without stronger connections among soil indicators, processes and services, the links between soil health, changed practice, and flow-on benefits, are currently not sequential nor clearly consequential. *Note*: 'Environment' includes climate, parent material, topography, native vegetation, and time.

1997), which are then used to assess soil health. As such, the soil health concept is entrenched in a properties-indicators-health 'loop' (Fig. 1) that is restricted to agricultural production potential, rather than to a broader capacity for a range of ecosystem services. Without stronger integration of ecological soil indicators, including tested relationships with a wide range of processes and services, soil health may remain a somewhat nebulous concept that has little direct relevance to contemporary visions of soil management for both agricultural and non-agricultural benefits.

4. Mechanisms for delivering public benefits: soil-based ecosystem services

4.1. Benefits produced from services

As outlined above, this paper's context involves changes in public benefits that arise from public interventions designed to encourage change in soil management on private land. In this context, public benefits are the net benefits '. . . accruing to everyone other than the private land manager [from that public intervention]' (Pannell, 2008). They involve broad societal benefits (e.g. clean air), rather than benefits accruing to individual members of the public (e.g. cheaper vegetables). In addition, net benefits exclude intervention costs borne by the public (e.g. payments) – so that relative benefits per intervention costs can be assessed (Pannell, 2008) – but include broad societal costs (e.g. dirty water).

Increasingly, public benefits from natural resource management are discussed in terms of ecosystem services. Indeed, the prevailing definition of ecosystem services equates the two, i.e. 'ecosystem services are the benefits people obtain from ecosystems' (Millennium Ecosystem Assessment, 2005). Under this definition, ecosystem services are provisioning (e.g. food, water, fiber), regulating (of, e.g. climate, floods, disease), cultural (e.g. recreation, aesthetics), or supporting (e.g. soil formation, nutrient cycling; Millennium Ecosystem Assessment, 2005).

Recent debate over the nomenclature of ecosystem services (e.g. Costanza, 2008; Wallace, 2008), has led to recommendations that the social purpose or decision context of a policy question should dictate the choice of service classification systems (Fisher et al., 2009). In this paper's context of relative valuation of changed practice to justify public expenditure (Fig. 1), Fisher et al. (2009) recommend keeping services and benefits separate, principally because multiple services can contribute to the same benefit, and that only benefits should be aggregated in valuation exercises to avoid double counting. They redefine ecosystem services as ecological phenomena that are 'utilized (actively or passively) to produce human well-being.' Benefits are then produced from services and are 'the point at which human welfare is directly affected' (Fisher et al., 2009). This classification also acknowledges that the same service (e.g. water flow regulation) can be utilized to produce multiple benefits (e.g. water volume, protection of physical assets), which can then be added together (Fisher et al., 2008). It provides a clear basis for decision-making, since, for example, it is simpler to decide between benefits as potential endpoints rather than to decide between the means of delivering those endpoints, i.e. services (Wallace, 2007).

4.2. Defining soil-based ecosystem services

Despite the currency of the ecosystem services concept, there are few clear and comprehensive definitions of soil-based ecosystem services. This has led to slack use of the service term in the soil context, and to confusion with related terms like processes (Wallace, 2007). Soil-based ecosystem services relevant to the public benefits context in Australia are listed in Table 1. Here, following a recommendation by Fisher et al. (2008), the focus is on 'final' services because these are directly utilized by humans. Final services are supported by single or multiple 'intermediate' services (final column, Table 1), which maintain the soil capital, but are not directly utilized (Fisher et al., 2008). This division of final and intermediate services is consistent with the Millennium Ecosystem Assessment, in which supporting (intermediate) services were defined as indirect, and were excluded from assessments of use and condition trends to avoid double counting (Millennium Ecosystem Assessment, 2005).

Consistent with Fisher et al. (2009), ecosystem services are defined as processes that become services if there are humans that benefit from them (Fig. 1). This interpretation is consistent with contemporary soil perspectives that soil-based services are essentially aggregates of soil processes (Palm et al., 2007; Kibblewhite et al., 2008), where processes are '. . . inputs, losses, and transfers of material and energy' (Palm et al., 2007). While a full listing of processes is beyond the scope of this paper, intermediate services and examples of associated processes (in brackets) are: 'soil structure maintenance' (aggregation, bioturbation, cheluviation); 'organic matter cycling' (litter comminution, decomposition, humification); 'nutrient cycling' (mineral weathering, mineralization, nitrification); 'ion retention and exchange' (cation exchange, anion adsorption); 'water cycling' (infiltration, evaporation, percolation, groundwater flow); 'gas cycling' (respiration, diffusion, denitrification, nitrogen fixation, methanogenesis); and 'soil biological life cycles' (changes in biotic richness and composition). In turn, a soil's capacity for processes is largely governed by key properties (Fig. 1; Carter et al., 1997). For example, Palm et al. (2007) identify texture, mineralogy, and soil organic matter as core properties that determine secondary soil properties (e.g. pH, bulk density, nutrient concentrations, aggregate stability), which together determine or constrain key process rates.

If 'service' indicates a positive outcome in the form of a benefit, then 'disservice' can be used to indicate a negative outcome or cost. Disservices reflect adverse changes in processes and intermediate services that are manifested as soil degradation. They are listed in Table 1 as the most common forms of soil degradation recognized in Australia, but can equally be interpreted as a loss in capacity to provide particular services (e.g. decreased capacity for disease and pest regulation). Here, Palm et al. (2007) note that while soil degradation is very familiar, the underlying mechanisms of degradation in terms of relationships and thresholds among soil properties, processes and (dis)services remain under-studied.

4.3. Linking services to benefits

Public benefits potentially arising from changed soil management are linked with soil-based final services or disservices in Table 2. It is immediately obvious that many of the services contribute to more than one benefit, and that individual benefits are often the product of more than one service. For example, the service 'soil structure stabilization' is central to, but not solely responsible for, a range of benefits including future choices, clean air, water quality, protection of physical assets, and ecosystem resilience (Table 2). As noted above, this 'joint production' of benefits by services illustrates the potential dangers of double counting if services rather than benefits are valued in environmental decision making (Fisher et al., 2009). In addition, the separation of benefits from services is warranted on the grounds that most members of the public will have greater experience and understanding of benefits, like favorable climate, than of underlying services, like gas regulation and carbon sequestration. Thus, they will have more tangible grounds for valuing benefits than services (Barkmann et al., 2008).

The most tangible examples of public benefits from soil management come from experience of soil-based disservices. For example, links between soil conservation and clean water have long been recognized, because negative soil–water interactions have contributed to the collapse of numerous societies in human history (Neary et al., 2009). Equally, the negative effects of wind erosion on air quality are very familiar (Doran et al., 1996). These conspicuous costs of poor soil management have provided a clear basis for soil monitoring programs in Australia, namely, the change in severity and extent of the most common forms of soil degradation (e.g. McKenzie and Dixon, 2006; VCMC, 2007). In turn, these data have utility in estimating mitigation benefits in payment incentive schemes (e.g. Eigenraam et al., 2007; Hajkowicz et al., 2008). Nonetheless, a number of potential public benefits in Table 2 could not obviously be linked with the amelioration of common disservices (e.g. novel products, disease and pest control, reduced pesticide use). Thus, purely focusing on the most obvious disservices could risk perpetuating the problem of ignoring and undervaluing many of the benefits (rather than just the avoided costs) produced by sound environmental management (Costanza et al., 1997).

Table 1
Soil-based ecosystem services and disservices appropriate to Australia (original list based on cited references). Intermediate services support final services, or their degradation is associated with final disservices. Disservices are listed here as the most common forms of soil degradation in Australia; alternatively, they can be the inverse of a final service. Codes link final services and disservices to public benefits in Table 2. Intermediate service abbreviations: 'SSM' soil structure maintenance; 'OC' organic matter cycling; 'NC' nutrient cycling; 'IE' ion retention and exchange; 'WC' water cycling; 'GC' gas cycling; 'BC' soil biological life cycles.

Code	Final services or disservices	Description	Intermediate services
	Final services (lead to benefits)		*Supporting*
S1	Provision of marketable goods	Provision of, e.g. food, fiber, timber	SSM, OC, NC, IE, WC, GC, BC
S2	Soil structure stabilization	Retention of soil (prevention of loss by wind and water)	SSM, OC, BC
S3	Gas regulation	Consumption/emission of atmospheric gases	SSM, OC, NC, IE, GC, BC
S4	Carbon sequestration	Net carbon stored in soil	SSM, OC, NC, GC, BC
S5	Water quality regulation	Water filtration/purification	SSM, OC, NC, IE, WC, BC
S6	Water yield	Water storage and availability	SSM, OC, WC
S7	Water flow regulation	Mitigation of, e.g. runoff, flooding	SSM, WC
S8	Weather regulation	Ameliorate daily extremes in air temperature and moisture	OC, WC
S9	Remediation of wastes and pollutants	Breakdown, immobilization, or detoxification of excess or harmful organic and inorganic materials	OC, NC, IE, BC
S10	Disease and pest regulation	Control of potential pests and pathogens	BC
S11	Habitat provision/genetic resource maintenance	Habitat for and maintenance of soil biodiversity (genes, species, phyla, functional groups)	SSM, OC, NC, WC, GC
	Final disservices (lead to costs)		*Degrading*
D1	Salinization	Increase in soil soluble salt content (to levels that produce costs)	SSM, IE, WC, BC
D2	Acidification	Increase in soil acidity (to levels that produce costs)	SSM, IE, BC
D3	Wind erosion	Loss of soil by wind (to levels that produce costs)	SSM, OC, NC, IE, WC, GC, BC
D4	Water erosion	Loss of soil by water (to levels that produce costs)	SSM, OC, NC, IE, WC, GC, BC
D5	Organic matter decline	Decrease in soil organic matter content (to levels that produce costs)	SSM, OC, NC, IE, WC, BC

References: Carter et al. (1997), Daily et al. (1997), Swift et al. (2004), Van der Putten et al. (2004), Wall et al. (2004); Millennium Ecosystem Assessment (2005), Farber et al. (2006), Lavelle et al. (2006), Barrios (2007), Palm et al. (2007), Wallace (2007), Fisher et al. (2008), Kibblewhite et al. (2008), and Fisher et al. (2009).

5. Information requirements for public investment in soil management

5.1. Estimating change in public and private net benefit: a case study

As stated above, expenditure of public funds to encourage change in soil management on private land should be based on evidence that public benefits outweigh costs; that is, evidence of public net benefits. In addition, private net benefits also require consideration because these will greatly influence the likelihood that private landholders will adopt the proposed change (Pannell et al., 2006). Relative changes in public and private net benefits can also help identify the most appropriate policy mechanism for fostering change. For example, extension or community outreach is a sensible policy tool where a proposed change is perceived to deliver strong net benefits to both the private landholders and the public, whereas positive incentives are likely to be more appropriate where public net benefits are strongly positive but private net benefits are minimal (Pannell, 2008). As such, estimation of public to private net benefits is considered a core component of newly emerging frameworks for public investment in natural resource management across southern Australia (Roberts and Pannell, 2009).

Fig. 2 summarises case study links between a change in soil management and potential changes in both public and private benefits and costs. The case study involves introduction of conservation tillage practices on light-textured Calcarosols (Isbell, 2002), a predominant soil type of the Murray Mallee Bioregion of north-west Victoria, Australia. This is a semi-arid region characterised by low relief dune fields formed from aeolian deposits of the late Pleistocene that typically carry mallee (*Eucalyptus* spp.) shrublands and woodlands (Gibbons and Rowan, 1992). The region has been extensively cleared for agriculture, which is predominantly dryland cereal cropping (DNRE, 2001). The most common form of crop preparation is 'conventional tillage', which typically involves a fallow/wheat/pasture rotation, with multiple tillage operations

Table 2
Public benefits potentially impacted by changes in soil management. Benefits are produced from increases in associated services or decreases in disservices (codes and references as per Table 1).

Public benefit	Description	Service	Disservice
Rural economic activity	Decreased vulnerability of rural societies	S1	All
Future choices	Sustained soil capital to accommodate future land uses or expectations	S2, S9, S10, S11	All
Clean air	Healthy air quality (e.g. low dust load, low pollutants)	S2, S3, S9	D3, D4
Favorable climate	Climate change mitigation, and local climate amelioration	S3, S4, S8	D3
Water quality	Water quality meets or exceeds standards for required uses	S2, S5, S9, S10	All
Water volume	Sufficient quantity of water available for required uses	S6, S7	D2, D3, D4
Protection of physical assets	Protection of buildings, machinery, etc. against, e.g. excess windborne soil, landslide, flood damage	S2, S5, S7	D1, D2, D3, D4
Novel products	Discovery/development of new public good products for, e.g. pharmaceuticals, material development	S11	–
Pollution control	Containment of wastes, pollutants, toxins	S9	D2
Disease and pest control	Containment of soil-based diseases and pests	S10	–
Reduced pesticide use	Reduced exposure to potentially harmful chemicals	S10, S11	–
Soil inoculation potential	Increased potential for inoculation by useful biota (e.g. root symbionts in revegetation)	S11	–
Ecosystem resilience	'Insurance' (and associated avoided cost) for disturbance recovery in the form of, e.g. stored water, functional diversity of biota	S2, S4, S6, S11	All
Aesthetics	Expectations of soil-based aesthetics, sense of place, cultural heritage	S2	D1, D2, D3, D4

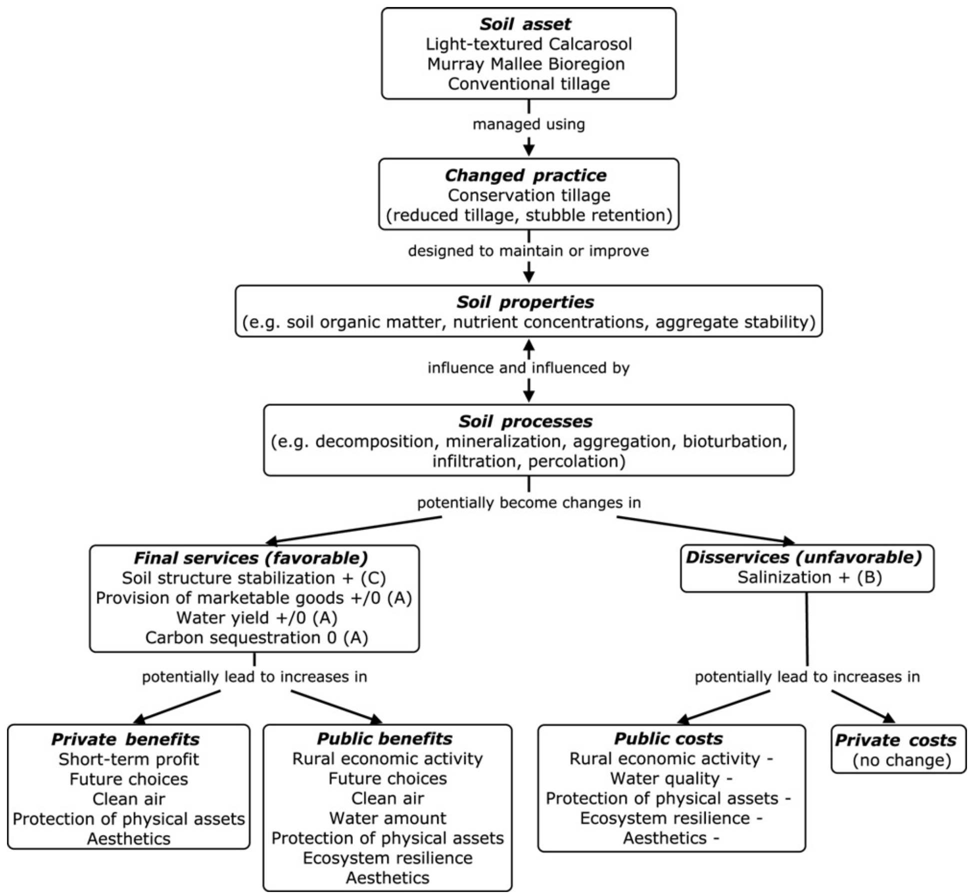

Fig. 2. Linking change in management of a soil asset (light-textured Calcarosol in the Murray Mallee Bioregion, north-west Victoria, Australia) with information required to assess the case for public intervention. Improved soil management is designed to improve soil properties, which both influence and are influenced by soil processes. Changes in soil processes become either favorable changes in final services that lead to benefits, or unfavorable changes in disservices that lead to costs ('–' indicates a decrease). Deciding if public expenditure is warranted to encourage the change requires information on relative changes in both public and private benefits, and public and private costs. Services and disservices were only included where they were examined for this scenario in published literature ('+' consistent evidence of increase; '+/0'; evidence of both increase and no change; '0' consistent evidence of no change). Letters in brackets indicate the best available level of evidence: 'A' measured or modelled data relevant to this soil type and changed practice in this bioregion; 'B' no data but inference made for this soil type and changed practice in this bioregion; 'C' no data but inference made for any soil in this bioregion under this changed practice.

Sources: McTainsh et al. (1990), Bird et al. (1992), Incerti et al. (1993), O'Leary and Connor (1996), O'Leary and Connor (1997), O'Leary and Connor (1998), Chan et al. (2003), Latta and O'Leary (2003), O'Connell et al. (2003a), Díaz-Ambrona et al. (2005), and Vu et al. (2009).

during the long fallow period (ca. 10 months; Incerti et al., 1993; DNRE, 2001). However, cropping has consistently been associated with severe wind erosion in Victoria's Mallee (McTainsh et al., 1990), leading to government agency campaigns to promote 'conservation tillage' practices, including zero or reduced tillage (e.g. chemical fallow), and stubble retention (DNRE, 2001; Chan et al., 2003). It was expected that these measures would be most effective on soils with weakly structured surface horizons like the light-textured Calcarosols, which are also characterised by low organic carbon content, and are at risk of (dryland) salinity (Chan et al., 2003).

With the exception of provision of marketable goods, there was minimal published information about changes in services with conversion from conventional to conservation tillage in the case study area. This was perhaps not surprising in an agricultural landscape where most experimental evaluations were primarily designed to identify constraints to crop production. Nonetheless, Fig. 2 lists five services/disservices for which there was some relevant published information. With the exception of carbon sequestration, for which two studies indicated no response (Chan et al., 2003; Vu et al., 2009), the published evidence indicated potential for increases in soil structure stabilization, provision of marketable goods, water yield, and salinization with conversion to conservation tillage on these soils. Measured or modelled data were available for three services (provision of marketable goods, water yield, carbon sequestration), but evidence of the remaining two was inferred for the soil type (salinization), or for soils in the bioregion as a whole (soil structure stabilization; Fig. 2). Despite promotion of conservation tillage for its potential to reduce soil erosion (DNRE, 2001), there appeared to be no direct examination of the effects of conservation tillage on soil structure stabilization or erosion of this soil type in this bioregion. This is not to say that this evidence does not exist, just that it is currently lacking in peer-reviewed literature (so is inaccessible or of unknown quality). Similarly, greater likelihood of salinization was inferred for conservation than conventional tillage due to increases in water storage (from enhanced infiltration and reduced evaporation) and associated deep drainage, which poses risks of raising saline groundwater in semi-arid regions (O'Leary and Connor, 1997; O'Connell et al., 2003a; Díaz-Ambrona et al., 2005). However, the specifics of that disservice in the case study context remain under-examined. For example, sand dunes in the Mallee are reported to have local groundwater flow systems (Ridley and Pannell, 2005), suggesting mostly within-farm saline discharge (Pannell et al., 2001). However, inferences about salinity from conservation tillage studies on these soils assumed regional-scale impacts (O'Leary and Connor, 1997; Díaz-Ambrona et al.,

Table 3
Estimated change in public and private net benefits produced by a change in soil management of light-textured Calcarosols in the Murray Mallee Bioregion from conventional tillage to either conservation tillage or restored native vegetation. Estimates were mostly inferred from relative anticipated changes in both associated services and disservices (Table 2). Anticipated change of '+3' indicates considerable increase in net benefit, '0' indicates no change, and '−3' indicates considerable decrease in net benefit relative to conventional tillage ('ND' not determined due to insufficient information).

Net benefit type	Anticipated change (−3 to +3)	
	Conservation tillage	Restored
Public		
Rural economic activity	0	−2
Future choices	+1	+2
Clean air	+2	+3
Favorable climate	0	ND
Water quality	−1	+1
Water volume	+1	−2
Protection of physical assets	0	+2
Novel products	ND	ND
Pollution control	ND	ND
Disease and pest control	ND	ND
Reduced pesticide use	ND	ND
Soil inoculation potential	ND	ND
Ecosystem resilience	+1	+1
Aesthetics	+1	+1
Balance	+5	+6
Private		
Short-term profit	0	−2
Financial certainty	0	−1
Ease of implementation	0	−1
Future choices	+1	+1
Clean air	+2	+3
Protection of physical assets	+2	+3
Reduced pesticide use	ND	ND
Aesthetics	+1	+1
Balance	+6	+4

2005). Thus, in the absence of more detailed analyses, the latter interpretation was accepted and only off-site impacts of potentially increased salinization were assumed in estimates of public and private net benefits (below).

If evidence of most services was minimal, then evidence of public benefits from a change to conservation tillage in our case study was negligible. This meant that potential public benefits listed in Fig. 2 had to be largely inferred from the links established between services/disservices and benefits in Table 2. In turn, the anticipated level of change in public net benefit associated with change to conservation tillage could only be qualitatively assessed (Table 3). This approach has precedence in recent literature (Van der Putten et al., 2004; Farber et al., 2006), and is considered a worthy option where low information levels preclude detailed ecological-economic modelling (Farber et al., 2006). Interestingly, repeating the assessment for a markedly different land-use change, namely restoration of native vegetation, indicated a similar balance of public net benefit (Table 3). Here, benefit estimates were based on inferred evidence of increases in soil structure stabilization, and decreases in provision of marketable goods, water yield, and (regional) salinization from just five published references, most of which examined benefits from increasing areas of woody plants, rather than benefits from stringent restoration of native vegetation communities (Bird et al., 1992; Knight et al., 2002; Unkovich et al., 2003; Crossman and Bryan, 2009; Bryan et al., 2010).

Evidence of changes in private benefits and costs in the case study was predominantly from measures of crop yield. Three studies found no change (Incerti et al., 1993; O'Leary and Connor, 1996; Latta and O'Leary, 2003) and two found increases (O'Leary and Connor, 1998; O'Connell et al., 2003a) in crop yields with change from conventional to conservation tillage. Input costs were estimated to be both lower (Incerti et al., 1993) and higher (Vu et al., 2009) under conservation tillage. Thus, the available evidence suggested no appreciable change in private short-term profit with change to conservation tillage (Table 3). In contrast, an appreciable decrease in short-term profit could be expected with change from conventional tillage to restored native vegetation (Table 3), mainly as opportunity costs from foregone agriculture (Crossman and Bryan, 2009), and assuming no increases in crop prices due to less overall production (Fraser and Hone, 2003). However, private benefits are in principle broader than short-term profit and include a range of social and environmental considerations that influence the relative advantage of one land use over another (Pannell et al., 2006; Pannell, 2008). Thus, potential private benefits arising from management change could also include financial certainty (i.e. reduced financial risk due to established markets, proven technology, reliable production), and ease of implementation (familiar, convenient, low complexity), as well as on-site environmental benefits including, for example, future choices (based on sustained soil capital; Table 2), clean air, protection of physical assets, reduced pesticide use, and aesthetics. As for public net benefit, the levels of these additional private benefits/costs were largely unknown in the case study. On balance, the increase in private net benefit was estimated to be greater with change to conservation tillage than with restoration to native vegetation. This was mainly due to lower short-term profit under native vegetation, and to assumptions of decreased financial certainty and ease of implementation associated with this land use (Table 3). Nonetheless, the balance is highly subjective, and sensitive to even minor changes in individual benefits; for example, an increase in private aesthetics from 1 to 2 with restoration would make the two land-use changes comparable (private net benefit balance 6 versus 5; Table 3). This of course assumes that all benefits would be valued equally (more below).

5.2. *Qualitative assessment of net benefits: positives and negatives*

There are a number of positives associated with the qualitative approach used in Table 3 to assess net benefits from change in soil management. First, the approach is transparent and requires consideration of a range of potential benefits that might otherwise be overlooked in more implicit estimations. Second, it clearly identifies information requirements, including those benefits for which there is no reliable information ('not determined'; Table 3). Third, consideration of a range of benefits helps identify those that are jointly produced with private economic benefits, which can positively influence landholders' willingness to supply and to accept payment for the additional benefits (Wossink and Swinton, 2007). Finally, the approach recognises potential trade-offs between benefits – a reality that is 'acceptable and indeed inevitable in any managed landscape' (Kibblewhite et al., 2008).

The negatives associated with the approach used in Table 3 must also be acknowledged. First, while the estimates are obviously qualitative, they can imply confidence in the outcomes when many are uncertain and probably contestable. In particular, many estimates are based on a limited number of sub-paddock changes in soil properties (e.g. organic matter content, water content). Delivery of public benefits from these changes relies on many assumptions, including changes in associated processes and services (Figs. 1 and 2), and the scaling-up of these changes over areas large enough to produce utilizable benefits. Importantly, the area of changed land use required to produce detectable change in benefits is likely to vary between services/disservices; for example, appreciable reduction in regional wind erosion might be achieved by increasing the area of land under perennial vegetation by 5% (Bird et al., 1992), whereas a reduction in regional salinity would likely require increases in the area of perennial vegetation in excess of 50% (Pannell and Ewing, 2006). In addition, the vertical scale of interest will vary between services/disservices. That is, while soil surface assessments are relevant to many services (e.g. provision of marketable goods, carbon sequestration), problems of acidification, salinity, and elemental toxicity can arise at depth in many Australian soil types (see McKenzie et al., 2004), clearly indicating a need to examine the full soil profile.

Another difficulty with the qualitative assessment of net benefits illustrated in Table 3 is a facade of constancy in the many underlying relationships. Relationships between practices and service changes are likely to vary with climate, particularly in arid and semi-arid regions. For example, increases in soil water yield can be appreciable under stubble retention (versus bare soil) after medium rainfall, but negligible under low and high rainfall (Monzon et al., 2006). In addition, while practices like intensive cropping (versus long fallow) can reduce the potential for deep drainage and related salinity problems in wet years, they can also dramatically reduce water available for growth of the next crop in dry years (O'Connell et al., 2002; O'Connell et al., 2003b). This suggests changes in the form of relationships between services with climate that are yet to be fully explored (i.e. always jointly produced? always a trade-off?).

Finally, based on the balance of public and private net benefits, the approach in Table 3 offered low power for choosing between two quite different changes in soil management. Clearly, discrimination would increase if the values of the anticipated changes in net benefit were known. In its simplest form, this involves assigning qualitative value ranks to each benefit (e.g. water quality 3, aesthetics 1), multiplying these by the anticipated change in that benefit, and summing the products for each proposed management change (Farber et al., 2006). Value rankings could be made through expert judgements, or through landholder and community consultations (Farber et al., 2006). Alternatively, as indicated below, there are a number of economic methods for more quantitative assessments of benefit value (Chee, 2004). While acknowledging the need for benefit valuation, there was little information available for valuing benefits and costs in the case study (although see Bryan et al., in press regarding opportunity costs). It is also likely that the value of benefits will change over time, particularly under climate change; for example, costs associated with salinization in the Mallee are predicted to decrease under climate warming and drying, whereas those associated with wind erosion will likely increase (Bryan et al., 2010). Furthermore, as discussed below, there are a number of benefit criteria, in addition to economic value, that warrant consideration when choosing between soil management systems for public investment.

5.3. *Who benefits when? Criteria for distinguishing benefits in decision making*

Sizeable knowledge gaps mean that most decisions about public investment in soils cannot account for all potential public benefits. However, initial consideration of each benefit could at the very least help with identifying priority benefits in the planning process, to address questions like – what mix of benefits can be realistically influenced? And when and where are these benefits likely to be realized? In this sub-section, a number of criteria for distinguishing between public benefits in an investment context are examined. These criteria provide some structure to the early stages of the decision-making process, so that potentially hidden benefits are not overlooked, and that goals relating to benefits are both explicit and realistic. The criteria (likelihood, degree, consequence, scale, direction, time lag, valuation) and related codes are defined in Table 4, and are then broadly assigned to soil-related benefits in Table 5. Where possible, just one code was assigned per criterion and benefit, but in many cases a range of codes was more realistic given that the nature of any benefit will vary with both the environment and the socio-economics of a decision context.

'Likelihood' is the probability that a benefit will be produced, and can conceivably range from unlikely to highly likely for any benefit, depending on the context. 'Degree' is the size of the change in benefit, which could be quantitatively predicted but, in the absence of robust data, will often be estimated within a range from small to large (either positive or negative; Table 4). Together, likelihood and degree are a measure of the anticipated change in net benefit production from a public investment in soil management (Table 3). Benefit changes that are likely and detectable warrant the application of the remaining criteria (Tables 4 and 5).

The consequences of an increase or decrease in benefit production require consideration. At a general level, not all benefits will have the same impact on human well-being. For example, changes in the production of clean air and water can be considered more vital to human existence than changes in the production of aesthetics (high versus low consequence; Table 5). Clearly, estimation of consequence requires some value judgment, and distinctions between less conspicuous benefits might require more detailed

Table 4
Description of benefit criteria and related codes used in Table 5 (original list compiled from cited references).

Criteria	Description	Criteria range or codes
Likelihood	Probability of benefit being produced	Unlikely to high
Degree	Size of the change in benefit (positive or negative)	Small to large
Consequence	Overall importance of the benefit to human well-being (or severity of impact if benefit is not produced)	'L' low; 'M' medium; 'H' high (potentially severe consequences if benefit is not produced)
Scale	Distance from soil management change to benefit production	'O' on-site (<100 m, *in situ* delivery); 'L' local (off-site, 100 m–10 km); 'R' regional (10–1000 km); 'G' global (>1000 km); 'I' independent (does not depend on proximity)
Direction	Direction from soil management change to benefit production	'O' omni-directional (all directions, no bias); 'DWind' Wind directional (according to wind directions); 'DWater' Water directional (according to water flow);
Time lag	The earliest time to benefit production after the change in soil management	'I' immediate (<1 year); 'F' fast (1 year to ≤10 years); 'M' medium (11 to ≤30 years); 'S' slow (31 to ≤50 years); 'VS' very slow (>50 years)
Valuation	Most feasible method for valuation of benefit	'M' market (priced using existing market); 'FM' future market (priced from future markets); 'CV' contingent valuation (simulated market based on e.g. willingness to pay for benefit); 'AC' avoidance cost (costs avoided if benefit is realized); 'RC' replacement cost (cost to replace or restore if benefit is not realized); 'TC' travel cost (price willing to pay to travel to benefit provision); 'H' hedonic (price willing to pay in related market, e.g. real-estate values for aesthetics)

References: Chee (2004), Lavelle et al. (2004), Ridley and Pannell (2005), Farber et al. (2006), McNeill and MacEwan (2007), Costanza (2008), and Fisher et al. (2009).

Table 5
Criteria for prioritising public benefits in public investment decisions. Criteria codes (described in Table 4) indicate the potential nature of each benefit across a broad range of contexts (i.e. assuming sufficient 'likelihood' and 'degree' to produce a detectable change in the benefit; Table 4). References as per Table 4.

Public benefit	Criteria				
	Consequence	Scale	Direction	Time lag	Valuation
Rural economic activity	H	L–G	O	F–M	M
Future choices	H	O–L	O	VS	FM
Clean air	H	O–G	DWind	I–F	CV, AC, RC
Favorable climate	H	L–G	O	F–VS	CV
Water quality	H	L–R	DWater	F–VS	M, RC
Water volume	H	L–R	DWater	M–VS	M
Protection physical assets	M, H	L–R	DWind, DWater	I–VS	AC
Novel products	L, M	I	O	VS	FM
Pollution control	H	O–R	DWind, DWater	F–S	CV, AC, RC
Disease and pest control	M, H	O–R	DWind, DWater	I–F	AC, RC
Reduced pesticide use	M	O–L	DWind	I–F	CV, AC
Soil inoculation potential	L	O–R	DWind, DWater	F–M	RC
Ecosystem resilience	H	O–R	O	M–VS	AC
Aesthetics	L	O–L	O	F–M	CV, TC, H

valuation (see below). However, consequence also represents the longer-term stability of benefit production, particularly flow-on effects to future generations if the benefit is not maintained in the present. For example, lower rural economic activity could lead to declines in rural population and infrastructure, thereby reducing capacity for future rural activity. Similarly, if the benefits of pollution control and ecosystem resilience are not maintained, then thresholds – either societal or biological – might be crossed that severely limit their potential for production in the future. Nonetheless, there remains 'fundamental uncertainty' about such thresholds in both ecosystem service provision and benefit production (Fisher et al., 2008).

Implicit in any discussion of public benefits is the question of scale – where will the benefits be produced? How many and which people will benefit? In particular, does the beneficiary catchment (local, regional, national) correspond with the spatial objectives of the investment decision? With some exceptions (e.g. novel products), benefits should be produced at a predictable distance from a change in soil management. Depending on the context, this distance could range from 'on-site' (<100 m) and 'local' (100 m to 10 km), to 'regional' (10–1000 km) or 'global' (>1000 km; Tables 4 and 5). In some cases, the scale of change in benefit will correspond with the scale of change in associated services. For example, soil structure stabilization is an on-site service that can be utilized to produce an on-site benefit of future choices. However, that same service can produce benefits at much broader scales, including water quality at regional scales (e.g. Lant et al., 2005), and clean air at regional to global scales (Leys et al., 2009).

Where benefits are produced will also depend on directional relationships between services and benefits. A service like soil carbon sequestration, for example, can produce a benefit of favorable climate without directional bias, whereas, water yield will produce a benefit of water volume in the direction of water flow (Fisher et al., 2009). Equally, wind directions can determine the location of many benefits including clean air, and pollution control (Table 5). Understanding these relative distributions of services and benefits can help identify the most effective locations for management intervention (Fisher et al., 2009), as well as the optimal configuration of land uses across landscapes (Lant et al., 2005).

Time to realization of different benefits will also require consideration in many investment decisions. Soil resources can be quickly depleted through severe disturbance – for example, by intense erosion – leading to impacts on associated benefits (e.g. clean air, water quality) within weeks to years (immediate <1 year to fast, 1–10 years; Tables 4 and 5). While establishment of perennial cover might reliably reduce wind erosion and quickly lead to cleaner air (Bird et al., 1992), increases in soil carbon sequestration will likely take considerably longer (tens or hundreds of years; Lavelle et al., 2004), and the associated benefit, through favorable change in climate, longer still. Similarly, time lags in off-site benefits from salinity control can range from 10 to >100 years, depending on the system's conductivity to water and on the scale of groundwater flow systems (Ridley and Pannell, 2005). Thus, depending on the context, time lags to change in soil-based public benefits can range from immediate (<1 year) to very slow (>50 years; Tables 4 and 5). This then leads to the question – should distant benefits be given lower priority than immediate ones in contemporary decision-making? In an economic sense, discounting to present value can be used to compare benefits that occur at different times. However, this approach is controversial, particularly since even low discounting rates will substantially reduce future benefits relative to current costs (Pannell et al., 2001). As a compromise, Farber et al. (2006) suggest using different rates for different benefit times; for example, using market-determined rates on intra-generational (ca. <40 years) benefits, and zero to very low rates on more distant benefits (i.e. assuming contemporary stakeholders will be willing to support potential benefits of future generations).

An obvious criterion for prioritizing public benefits is their relative value, either real or perceived, to the current generation. As previously indicated, this is not the value of the whole benefit but of additional units of benefit, either gained or lost, from a change in soil management (Fisher et al., 2008). A full assessment of the many potential methods for estimating the economic value of benefit changes is beyond the scope of this paper (see Chee, 2004; Farber et al., 2006). Instead, Table 5 indicates the most feasible valuation methods for each soil-based benefit (as indicated by the cited references). In addition, the following should be considered in relation to value: (1) while pricing benefits with clear market value (rural economic activity, water volume) might be most expedient, their sole consideration will likely lead to a serious under-provision of public good (Fisher et al., 2008); (2) expressing value purely in monetary terms might not always be possible or desirable – value might also be expressed using more qualitative indices (e.g. happiness, vulnerability; Fisher et al., 2008); and (3) only those benefit changes identified as having priority by the preceding six criteria should require more detailed valuations in any decision-making context.

6. Final comments

As suggested by Campbell (2008), links between paddock-level soil management and broader public benefits remain underexamined in the Australian context. In particular, there is minimal information on connections between current indicator-based assessments of soil health and broader-scale provision of non-agricultural ecosystem services. Thus, while recent definitions of soil health emphasize 'continued delivery of other [non-production] ecosystem services' (Kibblewhite et al., 2008), the basis for assessing soil condition against this definition is currently lacking.

This paper's case study highlights several impediments to making an informed benefits-based case for public intervention in private soil management in Australia. Notable knowledge gaps relate to the form, constancy, and spatial arrangement of relationships among soil management, ecosystem services, and public benefits. There is a clear need for better maps of services (Naidoo et al., 2008; Haygarth and Ritz, 2009), that extend beyond on-site provision (e.g. Crossman and Bryan, 2009; Gaiser et al., 2009), and of associated benefits, particularly off-site benefits (e.g. Lant et al., 2005). There will be many instances where soil-based services do not occur in the same place or at the same time as the benefits they provide. In any given environmental setting, single benefit production, as well as interactions between benefits, will likely vary with the weather and with the configuration and intensity of land uses. More explicit testing of the what, when, where, and how of benefit distribution is clearly required if the real public benefits arising from changed soil management on private land are ever to be fully appreciated.

Acknowledgements

This work was supported by the Land Health Unit (LHU), Natural Resources Division, of the Victorian Department of Sustainability and Environment (DSE). The authors would like to thank: Cate Turner and Dugal Wallace from LHU for ongoing support; Adam Hood (DSE), Anna Roberts and Mark Imhof (both Department of Primary Industries) for contributing to early discussions; and all participants in a one-day workshop on the public benefits of soil health.

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