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Effects of land markets and land management on ecosystem function: A framework for modelling exurban land-change☆

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ABSTRACT

This paper presents the conceptual design and application of a new land-change modelling framework that represents geographical, sociological, economic, and ecological aspects of a land system. The framework provides an overarching design that can be extended into specific model implementations to evaluate how policy, land-management preferences, and land-market dynamics affect (and are affected by) land-use and land-cover change patterns and subsequent carbon storage and flux. To demonstrate the framework, we implement a simple integration of a new agent-based model of exurban residential development and land-management decisions with the ecosystem process model BIOME-BGC. Using a stylized scenario, we evaluate the influence of different exurban residential-land-management strategies on carbon storage at the parcel level over a 48-year period from 1958 to 2005, simulating stocks of carbon in soil, litter, vegetation, and net primary productivity. Results show 1) residential parcels with management practices that only provided additions in the form of fertilizer and irrigation to turfgrass stored slightly more carbon than parcels that did not include management practices, 2) conducting no land-management strategy stored more carbon than implementing a strategy that included removals in the form of removing coarse woody debris from dense tree cover and litter from turfgrass, and 3) the removal practices modelled had a larger impact on total parcel carbon storage than our modelled additions. The degree of variation within the evaluated land-management practices was approximately 42,104 kg C storage on a 1.62 ha plot after 48 years, demonstrating the substantial effect that residential land-management practices can have on carbon storage.

1. Introduction

The effects of land-use and land-cover change (LULCC) on the carbon cycle and the provision of ecosystem services have become critical issues of global concern. Land-use and land-cover changes have contributed ~30% of historical anthropogenic efflux of CO_2, making them the second largest driver of anthropogenic CO_2 efflux, behind only fossil fuel burning (Sarmiento and Sundquist, 1992; Sundquist, 1993). Recent decades have seen rates of conversion of natural and agricultural land to low-density residential development exceed population growth in industrialized nations (Theobald, 2005). However, the processes and implications of LULCC on the carbon cycle and the provision of ecosystem services are not well understood.

The emerging field of land-change science addresses these issues (Rindfuss et al., 2004). Turner et al. (2007) identify four goals under which land-change scientists seek to improve our understanding: "(*i*) observation and monitoring of land changes underway throughout the world, (*ii*) understanding of these changes as a coupled human–environment system, (*iii*) spatially explicit modelling of land change, and (*iv*) assessments of system outcomes, such as vulnerability, resilience, or sustainability." This paper addresses the third point. Its primary goal is to present a new

☆ Thematic Issue on Spatial Agent-Based Models for Socio-Ecological Systems.

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agent-based modelling (ABM) framework that is designed to integrate biophysical, geographic, cultural and economic factors with land-use and land-management decision-making. Our secondary goal is to illustrate its performance by investigating the impacts of four land-management strategies on carbon balance dynamics in exurban residential land – a human-dominated ecosystem.

Few unified modelling efforts have explicitly incorporated the dynamic and combined effects of land-use change, land management, and land markets on LULCC patterns and ecosystem processes, especially in residential landscapes. Instead, most have focused on how individual mechanisms may affect LULCC patterns in isolation. For example, the influence of market mechanisms (e.g. Parker et al., 2012), demographics (e.g. Deadman et al., 2004; An et al., 2005; Fontaine and Rounsevell, 2009), and policy (e.g. Zellner et al., 2009; Robinson et al., 2009) on LULCC patterns are typically explored. These evaluations involve models that represent some processes explicitly and simplify others through the use of scenarios (e.g. Monticino et al., 2007; Kok and Van Delden, 2009; Verburg et al., 2010). Less frequently implemented are integrated models that combine multiple mechanisms to evaluate land-use change impacts at local, or continental to global scales (e.g. Schaldach et al., 2011; Schreinemachers and Berger, 2011).

Integrating multiple processes within the land system is necessary in order to understand how interactions among these processes produce complex behaviours across multiple outcomes of interest. For example, understanding impacts of LULCC on ecosystem services and social equity requires representation of ecological and socio-economic processes (Gaube et al., 2009). Our modelling framework is designed to facilitate evaluation of how policy, land-management preferences, and land-market dynamics affect LULCC and subsequent storage and flux of carbon (C) in exurban residential landscapes (Fig. 1). It can be used to explore a range of research questions, such as: How do land-management policies interact with social (neighbourhood/network), demographic, and economic factors to enable changes in land management that are likely to increase C storage? What are the relative impacts of initial land-cover patterns vs. changes in the frequency and intensity of land-management practices on ecosystem function? What is the relative effectiveness of planning vs. market-based policies in encouraging C storage in exurban landscapes? As a step towards answering these socio-ecological research questions, we constructed a simple scenario to evaluate the degree to which different land-management strategies may affect C storage.

We provide an overview and description of the design concepts for the framework components and how they interact, specific types of questions they have been designed to address, and outline the use of these protocols for various model implementations. Detailed descriptions of specific implementations are beyond the scope of this paper and reported elsewhere (e.g. Parker et al., 2012).

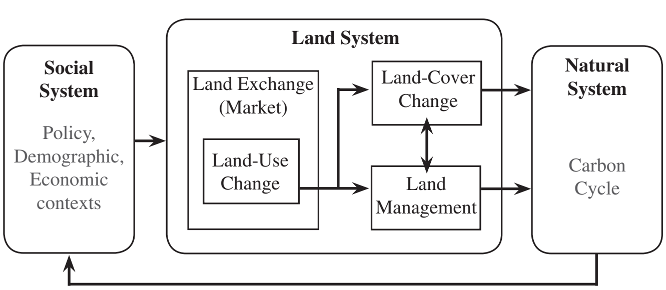

Fig. 1. Conceptual outline of the presented framework. Land-use change is shown as a component of land exchange because parcel exchange may occur without a change in land use. Likewise, market is placed in parentheses because land exchange may also occur in the presence or absence of market mechanisms.

2. The framework

The framework was developed within the empirical context of the exurban land system in Southeastern Michigan. Comprising patches of forest, grassland, turfgrass, buildings, and transportation infrastructure, the Southeastern Michigan landscape is typical of the highly fragmented human-dominated landscapes that are quickly expanding outside of typical urban and suburban areas within the United States and other developed countries Fig. 2.

The framework[1] comprises a collection of conceptual components that represent specific processes found in Southeastern Michigan and other exurban land systems. These components include the landscape, typology construction, agents, land-use change, the land market, land management, land-cover change, and the ecosystem impacts of land change, as measured through changes in rates and amounts of C storage in soils and vegetation. The components are grounded in previous modelling efforts that produced the following models: SOME and DEED (Brown et al., 2008; Robinson and Brown, 2009), ALMA (Filatova et al., 2009a), and Biome-BGC (Running and Hunt, 1993). Collectively the framework brings together the best components and conceptual approaches from these previous projects into an integrated approach for simulation and analysis.

Actors and their behaviours are represented as individual agents (see Section 2.2). These agents implement land-use changes across the landscape through their individual and collective land-exchange actions (see Section 2.3), while their specific attributes, preferences, and responses to socio-economic and policy contexts influence their decisions about land cover and land management (see Section 2.4). The local markets and exchange of information are modelled endogenously through agent interaction (Fig. 1) whereas the policy, demographic, and economic contexts of the social system are represented exogenously (Fig. 1). Changes to land cover and land management affect ecosystem function and the provision of ecosystem services in the exurban land system.

To demonstrate how the framework functions, we explore the degree to which exurban residential land-management affect C storage and flux at the parcel level. In this demonstration, we focus on the annual rates of C storage (or loss), together with decadal-scale changes in C storage, in exurban residential land as driven by agent choices and behaviours (see Section 2.5).

2.1. The landscape

The landscape comprises a collection of land-unit objects (cells, parcels, and subdivisions), which allows us to integrate the ecosystem model at several scales. Each land unit acts as a spatially explicit container for a set of biophysical and location-based state variables. Land units may include their own representation of natural processes or be used to send biophysical information to other models of natural-processes. They have hierarchical relationships that allow information to be passed in top-down or bottom-up directions (see Section 2.5).

2.2. Typologies

Our framework uses typologies to cluster similar objects, land uses, land covers, or agents into types with shared attributes, behaviours, or decision-making strategies. The variance in characteristics or structural components within a type is less than the

[1] The framework is implemented in Java, built using the Repast Simphony agent-based libraries within the Eclipse integrated development environment (Howe et al., 2006).

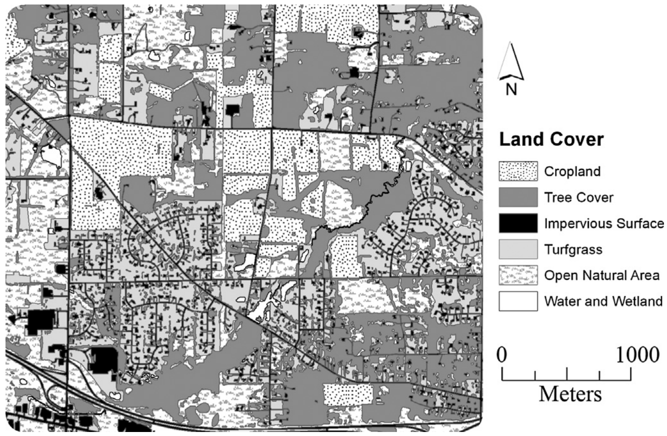

Fig. 2. Mosaic of land cover typically found across Southeastern Michigan.

variance between types. We can then observe and analyse types as wholes, simplifying the challenge of understanding complex system behaviour. Typologies have been used in previous ABM to reduce the detail required to represent a system (e.g. Smajgl et al., 2011), but care must be taken to ensure that reduced representations adequately represent system heterogeneity.

A typology may be strict in its classification (e.g. qualitative nominal groupings), nonstrict (e.g. quantitative values for attributes that may overlap among groups), or some mixture of the two. It may be based on expert opinion (e.g. Zellner et al., 2009), theory, or derived by data reduction techniques like cluster analysis (e.g. Fernandez et al., 2005; Fontaine and Rounsevell, 2009). An agent type may also be defined by an event or change of context (e.g. household life-cycle stages – Deadman et al., 2004; An et al., 2005) or an agent may belong to several different types (e.g. Huigen, 2004).

We defined four agent types that represent the key actors that drive the exurban land system: rural landowners (e.g. farmers), developers, land brokers, and residential households. The types have, in some instances, their own sub-typologies that generalize groups of agents based on the heterogeneity in their attributes, preferences, or behaviours. Section 2.3 provides a general description of these agents.

2.3. Agents

Rural landowners use the land to support a traditional livelihood (e.g. through agriculture). At any point in time it may decide to list a portion or all of one or more parcels that it owns for sale on the land market. The rural landowner's behaviours do not change until an offer is made that exceeds its willingness to accept price (WTA). When this occurs the landowner may sell the parcel.

Developers acquire land parcels to subdivide or aggregate for sale and profit. Each developer has its own preferences for the biophysical and geographical characteristics of a parcel or set of parcels. Developers influence land-management strategies on the parcels they create by initializing the proportions of land covers within them. When a parcel is acquired, a development template is applied that represents the land-use and land-cover patterns resulting from the development process. Each developer has a set of one or more templates that have the same dimensions as the parcel area. Typically template selection is based upon parcel size; however, a developer may have a set of templates that allow it to accommodate various policy or preference constraints that affect the type of developments that may occur.

Land Brokers facilitate the exchange of land between two agents by 1) providing information to agents concerning available parcels, 2) transferring property ownership, 3) posting sale opportunities and closures to the market institution, and 4) implementing negotiation mechanisms to determine the sale price among other functional activities.

Residential households participate in land exchange when entering the model at an exogenously defined in-migration rate, or when they attempt to move from one parcel to another within the model landscape. Each residential-household agent (RHA) seeks to find a settlement location that optimizes its utility under budget and informational constraints. If the RHA is successful in acquiring a parcel then it begins to manage the land within the parcel boundary (see Section 2.4). RHAs may also offer their properties up for land exchange. When an RHA determines to sell its parcel it may: 1) search out an alternative location that provides a higher utility and attempt to purchase that location before selling the current location; or 2) sell to the highest bidder and either a) exit the model, or b) look for an alternative settlement location. Land-management strategies and sale prices formed by RHAs may be a function of spatial neighbourhood or social factors (e.g. networks or norms).

2.4. Land-use change

We define land-use change as the exchange of land between two agents performing different land-use activities (e.g. residential or rural land use). Land exchange may, but does not necessarily, change land use (e.g. a property is transferred from one residential household to another) or land price. The exchange may occur in the absence (see Section 2.4.1) or presence (see Section 2.4.2) of a land market, as influenced by exogenous policy, economic, and demographic contexts (see Section 2.7). Land is supplied and may be demanded by rural landowner agents (e.g. farmers), subdivision developer agents, or RHAs.

2.4.1. The non-market approach

The framework provides the ability to model the exchange of land and its subsequent change in use without the inclusion of land-market processes. In the non-market approach, land policies provide the only constraints to land acquisition (e.g. Robinson and Brown, 2009). Non-market implementations of our framework focus on evaluating the effects of RHA preferences on geographical (e.g. nearness to water or roads) and biophysical (e.g. soil conditions, elevation, percent tree cover) characteristics on development patterns.

2.4.2. The land market

The land-market model borrows from the previously developed ALMA model (Filatova et al., 2009a, 2009b) and utilizes concepts from urban economics (e.g. location decisions under budget constraints, strategic bidding, and competition) to represent interactions between the demand and supply of land. When implemented within a model run, the land market is formed by bilateral trades amongst agents that buy and sell parcels. The collective outcome of these decentralized trades replaces traditional equilibrium price determination mechanisms (Arthur et al., 1997; Tesfatsion and Judd, 2006). The land market allows us to ask questions about the effects of interactions between market factors (e.g. credit availability, competition) and heterogeneous individual characteristics (e.g. budget constraints, strategic behaviour) on LUCC and ecosystem function.

Demand and supply. The supply of land (i.e. parcels) by an agent for purchase or acquisition by another agent is a function of policy constraints (e.g. zoning), existing stock of available land, and each agent's motivation for supply, willingness to accept (WTA) and ask prices. The rate of in-migration and existing agents' desires to relocate within or outside the region determines the aggregate regional demand. At the individual level, an agent's willingness to pay (WTP) is a function of transportation costs, parcel characteristics, location preferences, and budget constraints.

Preference and trade bundles. A buyer agent selects a parcel for acquisition based on its preferences for different location attributes, ranking parcels through a utility metric. To facilitate the use of a range of utility functions (e.g. linear, Cobb-Douglas, constant elasticity of substitution, and others) the framework uses a preference bundle object, which contains the relative attribute preference weights, the preferred value of those attributes, the observed values of those attributes (e.g. at a specific location), and an optional value representing the elasticity of substitution among the attributes. The framework can then incorporate any number of preference factors (e.g. accessibility, quality of school district) or empirical derivation of preference weights (e.g. Brown and Robinson, 2006).

Similarly, we define a trade bundle object that comprises the agent variables used in the processes of negotiation and land exchange, including but not limited to: a budget, WTA, WTP, agricultural reserve price, transportation cost per unit of distance, and the number of buyers and sellers in the market. Agents may then be parameterized to use one of several decision-making strategies to form ask or bid prices and to negotiate a sale price.

Market Transactions. Market transactions take place when two agents negotiate an exchange of land through a land-broker agent. Buyers form a WTP and then submit a bid to a broker agent or directly to the seller. The bid price is a function of WTP, excess supply or demand, and the seller's asking price. The seller, having already formed a WTA and posted an ask price, determines which bid (if any) to accept and may implement a price negotiation strategy. Differentiation between proposed prices (bid and ask prices) and reservation prices (WTP and WTA) allow us to model strategic market behaviour.

Following a sale, the seller submits transaction information to the broker agent, who then posts the sale to the market-institution object. This object tracks market information (e.g. number of buyers and sellers; recent transactions) and maintains a list of available parcels with characteristics for the broker agents to query. The market institution acts like a multiple listing service and facilitates the expansion of the framework to include new implementations of broker agents with specific behaviours (e.g. realtor agents).

2.5. Land management and land-cover change

We define a land-management strategy as the collection of land-management actions conducted by a landowner agent over a given year. Land-management actions may alter the proportion or quality of vegetated land-cover types and subsequent ecological and biogeochemical processes within a land unit. The choice of a land-management action or strategy can be based on the existing land management of a land unit, that of neighbours or other social contacts, expectations of land-market valuation, or agent attributes and preferences. Land-management actions may include mowing turfgrass, planting, cutting, or pruning trees, raking or mowing fallen leaves, and fertilizing or watering turfgrass, among other activities.

The management strategy can be imposed on all land units of a given type (i.e. subdivision, parcel, and cell) or specified for any specific land unit. This code structure can represent, for instance a case where a developer creates a subdivision with a corresponding management plan implemented uniformly on all parcels within the subdivision and all cells within each parcel. Alternatively, it also allows each RHA to perform one type of management strategy on its parcel or a set of specific cells within its parcel (e.g. different management on the front yard vs. the back yard).

2.6. Measuring the ecosystem impacts of land-use change and land management

To estimate the ecosystem impact of exurban development and land-management practices, the framework provides an ecosystem-component object that links the ABM to ecosystem models. The object facilitates the conceptual mapping of agent behaviours to ecosystem parameters, site conditions, and variables. By linking the ABM to an ecosystem model we are able to evaluate the effects of exurban development, the land market, and land-management behaviours on the relative and absolute amount of estimated C storage and flux. This includes both methodological questions, e.g. how does the scale of representation of land management affect estimates of C storage, and substantive questions and issues, e.g. how climate variability and change may affect C storage under a range of management behaviours or how exurban development may be moulded or retrofitted to store more C in future decades.

The ecosystem-component object design is based on our decision to use the widely published ecosystem-process model BIOME-BGC to estimate changes in C storage and flux. BIOME-BGC simulates vegetation growth and changes in ecosystem C storage and flux in individual pools (e.g. root, stem, and canopy) as well as gross and net primary production (Running and Hunt, 1993) for site conditions at a specific point. We scale the point-specific estimates to the land unit of analysis and summarize results across all land units to estimate regional ecosystem output. The ABM and ecosystem components are loosely coupled, to allow independent development of software components by ecosystem and land-use modelling teams.

2.6.1. Conceptual aggregation of residential vegetation ecology (CARVE)

To accommodate the simulation of ecosystem impacts at different land-unit scales, the framework utilizes an object, named CARVE, capable of aggregating quantities of different land covers (or biomes), site conditions, and management practices at each land unit. Furthermore, the cell land-unit resolution is configurable such that CARVE may supply land-cover cells with a resolution of 30 m (e.g. Landsat data) or 250 m (e.g. MODIS) to an ecosystem model or aggregate cells to a coarser resolution. This flexibility permits the model to be parameterized at any scale. When run in the absence of land-management behaviours (as is done as a baseline in our example application, see Section 3.2), the framework produces results similar to those obtained from BIOME-BGC applications to non-managed lands (e.g. Thornton et al., 2002; Jung et al., 2007).

The incorporation of CARVE has two main advantages. First, CARVE allows us to analyse the effects of resolution of representation on model outcomes. For example, often homeowners apply different management practices to different areas of their parcel (Joan Nassauer, personal communication). In the front yard a residential household may prune trees, rake leaves and grass, which remove a substantial amount of C. In the back yard it may let vegetation grow naturally and deposit extracted biomass from the front yard. Our ability to configure the scale of representation and output improves our ability match outputs to observations and other model results for validation and comparative purposes.

Second, a land unit may contain multiple instances of the same land-cover type with the same land-management actions. In this case the aggregation of cells, by CARVE, to the parcel level provides no loss of information and increases the computational performance of the model.

2.7. Land policy, demographic, and economic contexts

One goal in extending LULCC models to include land markets and land-management behaviour and linking to an ecosystem model is to evaluate a number of land-use and land-management policies that could be enacted to help keep highly fragmented and human dominated exurban landscapes acting as C sinks. Land-policy scenarios are implemented exogenously in our framework by manipulating specific model implementation parameters. Policy options could include minimum lot-size zoning regulations, use of tax or transaction costs to promote high vs. low rates of development, or carbon-storage incentives such as C payments.

3. Simulating the impacts of residential land management on carbon storage

To demonstrate how the framework functions, we use a simple model based on a contrived scenario to evaluate the degree to which different residential land-management strategies influence C storage at the parcel level over a 48-year period from 1958 to 2005. We apply the model to a hypothetical and stylized landscape that is representative of LULCC patterns and environmental conditions in Southeastern Michigan. We simulate changes in stocks of C in soil, litter, vegetation, and their sums, as well as net primary productivity (NPP) due to changes in land management. These scenarios emphasize the relationship between land-management practices and environmental outcomes, while minimizing the role of the land-exchange process.

3.1. The application example narrative

The narrative begins with a parcel owned and farmed by a rural-landowner agent. Two residential-developer agents bid on the parcel and the highest bid above the asking price is awarded the sale. The winning developer agent subdivides the farm parcel into four residential parcels of the same size (i.e. 1.62 ha or ∼4 acres), shape (3 cells wide by 6 cells deep, cell resolution = 30 m), and the same amount and location of three land-cover types (impervious to represent the house and driveway, turfgrass, and DTC, Fig. 3). Soil within each parcel is uniform and the texture and soil depth of each cell is 31% sand, 47% silt, 22% clay, and 1.5 m in depth. These soil profile characteristics were extracted from Soil Survey Geographic (SSURGO) soils data and similar profiles can be found throughout Southeastern Michigan. Parcels differ in their proximity to urban amenities, but biophysical characteristics are held constant.

The winning developer agent places notification of available parcels for acquisition to the market-institution object. RHAs ask the land-broker agent to provide them with a list of available residential lots, which is extracted from the market institution. RHAs evaluate the available residential parcels based on their preferences for distance to urban amenities and open space, which are known factors in the residential location decision (e.g. Fernandez et al., 2005). RHAs then place a bid to purchase the parcel that maximizes their utility – within their budget constraint. In this narrative, RHAs use the following Cobb–Douglas utility function from Filatova et al. (2009a):

$$U = A^{\alpha} \cdot P^{\beta}$$

where A is an open-space amenity measured as the density of undeveloped land in the focal neighbourhood, P is the proximity to urban amenities (i.e. the normalized distance from an urban centre), and α and β are their respective preference weights such that $\alpha + \beta = 1$. The broker agent mediates the purchase negotiation between the RHA and developer; however, in this simple example the developer agent owning the parcel selects the highest bid, above the willingness to accept, for each parcel and the supply of four parcels clear within a single time step (i.e. 1 year). Each RHA settles in the parcel it purchased and begins managing the land

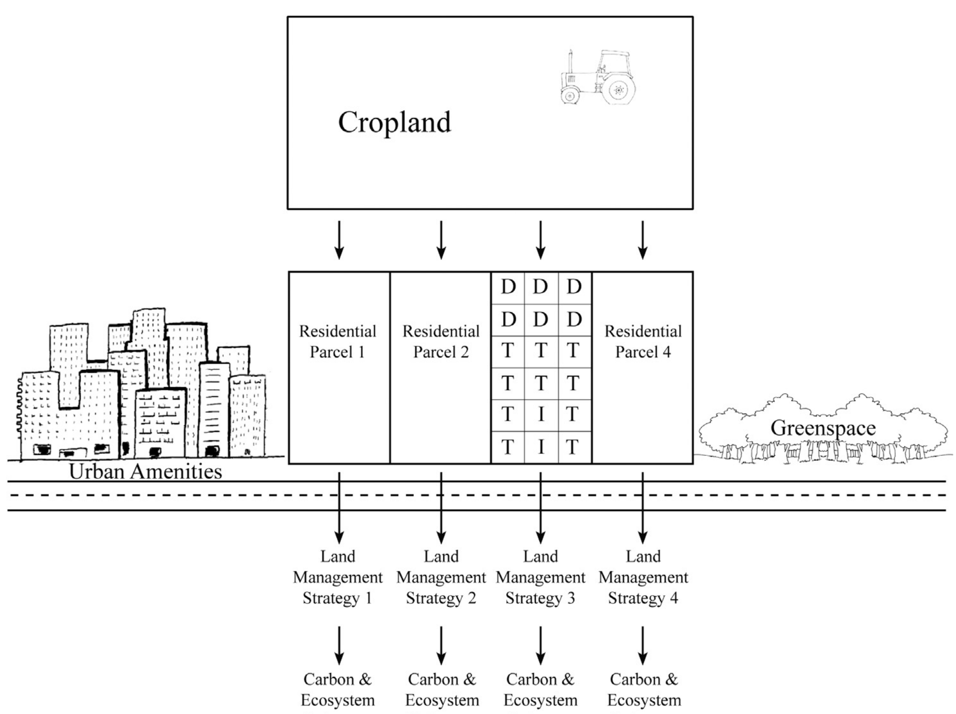

Fig. 3. Experimental design to evaluate the effect of different residential land-management strategies on C storage. D = dense tree cover, T = turfgrass, I = impervious surface. Distance from urban amenities (left) and greenspace (right) influence RHAs' evaluation of utility from residential parcels 1–4. Parcel 3 illustrates the land-cover pattern within each parcel.

based on a predefined land-management strategy. The land-management behaviour of the RHA is carried out each time step of the model (i.e. annual time steps) for the duration of the model run.

The modelling process just described can be generalized to any landscape composed of any number of parcels using the pseudo code in Fig. 4. We restrict our experiment to four parcels to show a parsimonious implementation of the framework and to illustrate the extent to which land management may affect C flux and storage in exurban landscapes.

3.2. Land-management strategies

We evaluate two types of land-management behaviours that affect C storage, those that partially remove nutrients and those that make additions to the system. From these two behaviours we construct four combinations (Strategies 1–4) to estimate the impacts of different land-management strategies on C storage (Fig. 5).

Removal strategies involve reducing litter C and N in turfgrass and coarse woody debris (CWD) in DTC to 1% of their existing levels at the end of the year. For the two addition strategies (2 and 4),

		Additions: No	Additions: Yes
Removals	No	**Strategy 1** No land management Baseline strategy for comparison. Model conforms to non-managed applications of BIOME-BGC	**Strategy 2** Fertilize and irrigate Hypothesis: upper boundary, expecting highest level of C storage and NPP
Removals	Yes	**Strategy 3** Remove litter from turfgrass and CWD from DTC Hypothesis: lower boundary, expecting lowest level of C storage and NPP	**Strategy 4** Fertilize, irrigate, and remove litter from turfgrass and CWD from DTC Hypothesis: expect levels of C storage and NPP between Strategies 2 and 3

Fig. 5. Land-management strategies performed by residential household agents (RHAs).

2.54 cm of water per week is applied to turfgrass (as recommended for maintenance by Schultz, 1999). We operationalize the process of irrigation by altering the precipitation values to include an addition of 2.54 cm of water once a week from May 1st to October 1st (22 applications). Also, fertilization is implemented on turfgrass and is operationalized as additions to the supply of soil mineral N. The

```
Start
Initialize the landscape, parcels, and owners, and exogenous processes
Spin-up BIOME_BGC for each location and adjust for land-use legacy effects
Model Step = 1 year
        Various resets for each step

        Interior Step
            Land Market
            Seller agents form ask prices
            If Developer agent is available to develop then
                Developer asks broker for available locations
                Developer examines locations and calculates utility
                Developer determines best location to develop
                Developer submits asking price to broker
                Broker negotiates sale
                Seller tells Broker to register sale
                Ownership exchange occurs
                Seller becomes buyer or leaves region
                Developer creates subdivision
                Developer submits new available parcels to broker
                Broker registers parcels with market institution
            Residential buyer agents ask broker for available locations
            Residential buyer agents examine locations and calculate utility
            Residential buyer agents determine and submit bid price(s) to broker
            Broker negotiates sale (or no sale occurs)
            Broker registers sale with market institution
            Seller becomes buyer or leaves region
        End Interior Step (when no more buyers exist)

        Land Management:  Agents implement strategies that apply additions to the landscape
        BIOME-BGC:        Step model forward one year
        Land Management:  Agents implement strategies that apply removals to the landscape

        Report Output (for the year)
End Model Step
Report Output (for the run)
End Model Step
```

Fig. 4. Process of actions in a typical implementation of the framework. Other implementations can replace various components, e.g. the Land market could be replaced by a simple Land exchange or BIOME-BGC could be replaced by a simpler submodel.

RHA applies an annual amount of 0.018 kg N m^{-2} yr^{-1}, which corresponds to the high range of fertilizer applications reported for turfgrass maintenance (Schultz, 1999) and with common products available on the market (e.g. Greenmaster Organic,[2] 2009 The Scotts Company LLC; Ultra Turf™ Turf Fertilizer 29-0-4,[3] Vigoro).

3.3. Initializing BIOME-BGC for exurban development and management

Ecosystem models typically estimate the potential vegetation for a location based on generalized parameter settings. To provide an approximate representation of exurban DTC growth in Southeastern Michigan, we calibrated BIOME-BGC by first running the model (in what is called the *spin-up process*) until a dynamic equilibrium among vegetation ecophysiology, nutrient pools and fluxes, and climate was met (Thornton et al., 2002). Once equilibrium was reached for presettlement deciduous broadleaf forest (DBF), we modified the equilibrium biogeochemical state variables to represent land-use change that occurred in the region (i.e. conversion to agriculture, which peaked in the late 1880s–1900). This involves removing above ground vegetation and altering below ground nutrient pools (see Robinson et al., 2009 for details, data used, and land-use history in the study region).

We create two ecophysiological parameter settings to represent the dominant land-cover types (i.e. DTC and turfgrass) in exurban residential. We simulate the growth of DTC for 77 years to produce model outputs of the same relative age as field-based observations. Model outcomes are compared to field-based observations of above-ground vegetation C (year 2006), total vegetation carbon, and below-ground soil C estimates (Table 1). The ecophysiological parameters representing DTC are the averages for a DBF as specified by White et al. (2000); (see Appendix A). However, to calibrate the model for a good fit with observed data we alter the fraction of leaf nitrogen in rubisco (variable FLNR in BIOME-BGC) from 0.33 to 0.07, which falls within the acceptable range of values for a parameter that is not well informed empirically (White et al., 2000; Robinson et al., 2009).

To represent turfgrass in exurban landscapes we first use the mean values for ecophysiological parameters representing C3 grasses as specified by White et al. (2000). We then modify those parameters as specified by Milesi et al. (2005); (see Appendix B) to represent turfgrass. We use the same spin-up process as was used for DTC and reduced the various nutrient pools to represent the clearing of the land of trees and its use in agriculture.

Similar to Tatarinov and Cienciala (2006), we increase the rate of annual whole-plant mortality for DTC. We alter the default value from 0.005 to 0.02 (i.e. a 2% mortality rate per year) to represent an increase in the mortality rate due to disease, wind-throw, and human-induced stressors that occur more frequently in human-dominated and fragmented landscapes. For turfgrass we reduced the rate of annual plant mortality to zero and for both turfgrass and DTC we set the annual mortality fraction due to annual fire events to zero, representing management of the property against fire events.

3.4. Integrating ABM and BIOME-BGC within the framework

Within a given annual time step, an RHA conducts its land-management actions within a given land unit. The land-management actions modify several input files used by BIOME-BGC. If the RHA irrigates, the ABM manipulates the climate file or selects the appropriate climate file to represent the additional contribution of irrigation to precipitation values. If the agent adds inputs like fertilizer to the land unit, the nutrient pools file (restart file) is modified (see additions section of Fig. 6). Lastly, the agent must update the time step in a file that describes the site characteristics and is used to initialize BIOME-BGC.

Once all of the agents have performed their behaviours, the ABM cycles through each land unit (in this case each cell within all parcels) and points BIOME-BGC to the appropriate input files. The ecosystem model runs for a single year and stores the state of the nutrient pools (see run BIOME-BGC in Fig. 6). The ABM then reads the nutrient pools and modifies them based on extractive land-management behaviours or transfer of nutrients from one pool or location to another (see removals Fig. 6). After these steps are completed the annual Total, Vegetation, Litter, and Soil C are output along with NPP in raster and tabular format.

[2] Applied 6 times per year from April 1st to September 1st at 45 g m^{-2} and 12% N content. Recommended application is calculated as 0.018 kg m^{-2} yr^{-1}.

[3] Applied 4 times per year from April 1st to September 1st at 16.15 g m^{-2} and 29% N content. Recommended application is calculated as 0.0187 kg m^{-2} yr^{-1}.

Table 1
Comparison of observed and modelled C storage values for DTC and turfgrass.

	Observed	Modelled
Dense tree cover		
[a]Above ground vegetation carbon (kg m^2)	9.13	11.51
Total vegetation carbon (kg m^2)	15.84	15.55
Soil carbon (kg m^2)	17.49	16.65
Turfgrass		
Above ground vegetation carbon (kg m^2)	0.14	0.09

All other values are the average of field measurements taken at 29 exurban residential locations in Southeastern Michigan (unpublished data).
[a] Field measurements and average literature values from Robinson et al. (2009).

4. Application example results

4.1. Dense tree cover (DTC)

The impact of residential land-management that removed coarse woody debris (CWD) over 48 years was a reduction in Litter (0.158), Soil (1.066) and an increase in Vegetation C of 0.212 kg C m^{-2} compared to the baseline residential parcel (Strategy 1) with equivalent land cover over the same period. Total C was 1.011 kg C m^{-2} lower with the removal of CWD (29.472 kg C m^{-2}) when compared to the baseline Strategy 1 (30.483 kg C m^{-2}, Table 2). Removal of CWD limited the amount of Soil C inputs and therefore, as we would expect, the simulated reduction in Soil C was the greatest among our three C pools, which comprised 87% of the loss. NPP was marginally higher with the removal of CWD, which lead to a slight increase in Vegetation C but not enough to offset Soil and Litter C losses.

4.2. Turfgrass

Management practices employed on turfgrass were more numerous than the simple removal of CWD employed on DTC areas. The addition of inputs (Strategy 2), led to an increase in C storage in all pools (except Litter) by 2005 relative to the non-managed lands (Strategy 1). These inputs provided an upper bound on Vegetation and Soil C storage in our experiments with each pool storing 0.206 and 18.681 kg C m^{-2}, respectively (Table 3). Strategy 2 Litter C was slightly lower (0.028 kg C m^{-2}) than Strategy 1 and NPP values were 0.034 kg C m^{-2} yr^{-1} higher with the inclusion of fertilization and irrigation compared to Strategy 1.

Consistent with expectations, Strategy 3 provided the lower bound of C storage and flux, as the residential parcel owner was annually removing litter from the turfgrass, with reductions of

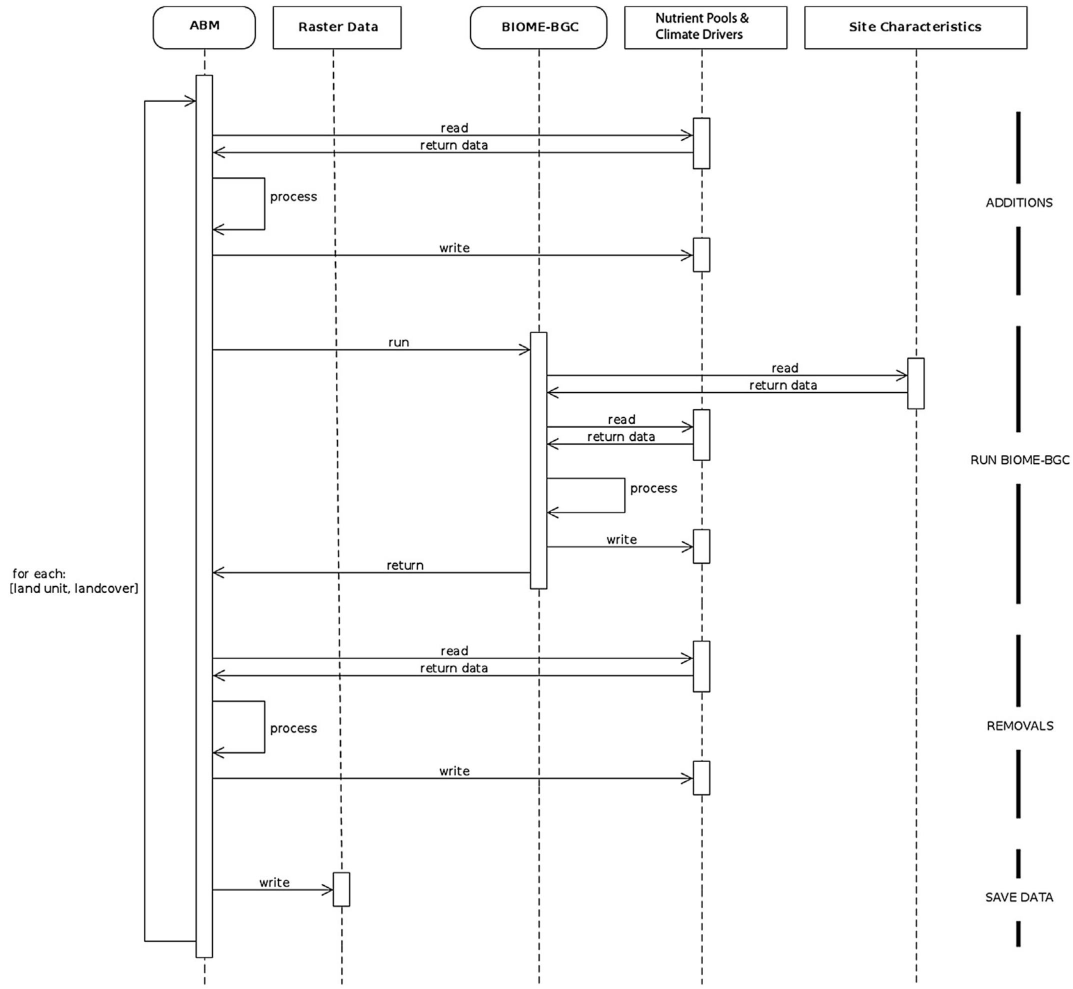

Fig. 6. The sequence of actions structuring the integration between the ABM framework and the ecosystem model BIOME-BGC.

Table 2
Simulated dense tree cover C storage and flux values for year 2005 in kg C m^{-2} (a). Proportion of C storage and flux relative to the baseline Strategy 1 (b). Note: The label removal refers to if an agent land-management strategy included nutrient removals (yes) of coarse woody debris or not (no). Additional inputs were not applied to DTC in the modelled land-management strategies; however the table is formatted for clarity based on the configuration of Fig. 5 and consistency of interpretation with Table 3.

Dense tree cover		(a)			(b)	
		Addition			Addition	
		No	Yes		No	Yes
Vegetation						
Removal	No	13.324	na	No	0.000	na
	Yes	13.536	na	Yes	0.212	na
Litter						
Removal	No	0.509	na	No	0.000	na
	Yes	0.351	na	Yes	−0.158	na
Soil						
Removal	No	16.650	na	No	0.000	na
	Yes	15.584	na	Yes	−1.066	na
Total C						
Removal	No	30.483	na	No	0.000	na
	Yes	29.472	na	Yes	−1.011	na
NPP						
Removal	No	0.581	na	No	0.000	na
	Yes	0.605	na	Yes	0.024	na

0.066, 0.588, and 3.238 kg C m^{-2} for Vegetation, Litter, and Soil C, respectively, relative to Strategy 1. Total C storage for Strategy 3 (15.196 kg C m^{-2}) was the lowest of the four land-management strategies. The two strategies (3 and 4) that included removals had substantially lower amounts of carbon storage for all pools and NPP, which suggests that C removals from the landscape have a greater impact than fertilizer and irrigation additions. Approximately 83% of the reduction in C storage from removals, relative to Strategy 1, was due to reduced Soil C storage. The removal of the primary C input to turfgrass soil substantially reduced the amount of C stored. These removals also led to a decrease in the annual accumulation of C, as shown by a simulated NPP of 0.247 kg C m^{-2} yr^{-1}, which is 0.132 kg C m^{-2} yr^{-1} lower than Strategy 1 and 0.166 kg C m^{-2} yr^{-1} lower than Strategy 2.

We did not know, *a priori*, whether Strategy 4 would have higher or lower C storage than the baseline Strategy 1, because we did not know whether removals would dominate additions. Given that removals had more effect than additions, we expected Strategy 4 to be lower than baseline (Strategy 1), which it is for all measures. We also expected that Strategy 4 (additions and removals) would be (slightly) higher than Strategy 3 (removals only). However a surprising result was that NPP and C storage pools were only slightly higher for Strategy 4 than for Strategy 3. Strategy 4 Vegetation, Litter, and Soil C values were only 3.758E-05, 5.485E-07, and 3.167E-03 higher than Strategy 3, respectively. Likewise, NPP was only 7.517E-05 kg C m^{-2} yr^{-1} higher for Strategy 4 than Strategy 3.

Table 3
Simulated Turfgrass C storage and flux for year 2005 in kg C m^{-2} (a). Proportion of C storage and flux relative to the baseline Strategy 1 (b). Note: The label removal refers to if an agent land-management strategy included nutrient removals (yes) of litter or not (no). The label addition refers to if an agent land-management strategy that applied nitrogen fertilizer and irrigation (yes) or not (no). The configuration of the table is based on Fig. 5 for each C pool or flux.

Turfgrass		(a)			(b)	
		Addition			Addition	
		No	Yes		No	Yes
Vegetation						
Removal	No	0.189	0.206	No	0.000	0.017
	Yes	0.123	0.123	Yes	−0.066	−0.066
Litter						
Removal	No	0.590	0.562	No	0.000	−0.028
	Yes	0.002	0.002	Yes	−0.588	−0.588
Soil						
Removal	No	18.309	18.681	No	0.000	0.372
	Yes	15.070	15.074	Yes	−3.238	−3.235
Total C						
Removal	No	19.089	19.449	No	0.000	0.361
	Yes	15.196	15.199	Yes	−3.893	−3.890
NPP						
Removal	No	0.379	0.413	No	0.000	0.034
	Yes	0.247	0.247	Yes	−0.132	−0.132

These results further emphasize our finding that our land-management practices that remove C have a stronger effect than the land-management practices that provide additions.

4.3. Parcel C dynamics

By scaling the above results to the parcel level (Sections 4.1 and 4.2) we begin to see the impact that different land-cover types and exurban residential land-management strategies can have on C storage. Each of the four residential parcels were 1.62 ha (~4 acres) with 33% (0.54 ha) in DTC, 56% (0.9 ha) in turfgrass, and 11% (0.18 ha) in impervious surface. Despite DTC comprising only 33% of the total parcel area, in the non-managed parcel DTC stored 164,608 kg C in year 2005, which is 49% of the Total C on the parcel.

Our results demonstrate that both the types of land cover on an exurban residential parcel and the management strategy employed can affect C storage. The C storage in one m^2 of DTC is initially very similar to turfgrass, with a ratio ~1.19; however, the ratio increases over time to 1.60, 1.57, 1.94, and 1.94 for Strategies 1–4, respectively. This C-storage ratio increased in the unmanaged parcel because the NPP of the larger and older-aged trees comprising DTC was over one and a half times that of turfgrass after 48 years. When land-management Strategies 3 and 4 were employed the ratio increased further because C was utilized for turfgrass growth and then removed from the system via litter removals. In contrast the removal of CWD in DTC had little effect.

The difference in the effect of different management strategies on C storage within a single exurban residential parcel is large (Fig. 7). The ranking by total parcel C storage of the management strategies we employed was additions only > no management > additions and removals > removals only (Table 4). These results suggest that 1) residential parcels with management practices that only provide additions in the form of fertilizer and irrigation to turfgrass store only slightly more carbon than parcels with no management, 2) conducting no land-management strategy stores more carbon than implementing a strategy that includes removals in the form of removing coarse woody debris from DTC and litter from turfgrass, and 3) the modelled removal practices have a larger impact on total parcel carbon storage than the modelled additions. What is most noticeable is that the degree of variation within the evaluated land-management practices was approximately 42,104 kg C storage on a 1.62 ha plot after 48 years, demonstrating the substantial effect that residential land-management practices can have on carbon storage.

Table 4
Changes in total parcel carbon from 1957 to 2005 (kg C) for each land management strategy. (d) Difference is relative to baseline management Strategy 1 of no management.

		(a) Total parcel C 1957			(b) Total parcel C 2005	
		Addition			Addition	
		No	Yes		No	Yes
Removal	No	285597.451	285597.451	No	336405.251	339651.281
	Yes	285597.451	285597.451	Yes	295909.983	295938.828
		(c) C Accumulation 1957–2005			(d) Difference in Total Parcel C 2005	
Removal	No	50807.800	54053.830	No	0.000	3246.030
	Yes	10312.532	10341.377	Yes	−40495.268	−40466.423

5. Discussion

5.1. Carbon storage in exurban residential landscapes

By investigating the C impacts of four land-management strategies, we first demonstrated that all exurban residential land managers increase C stored within their parcel area when the previous land use is cropland. The minimum amount of additional carbon stored was estimated to be 10,312 kg C after 48 years. Such a result is important given the fact that the rate of growth of exurban residential lands is outpacing population growth by approximately 25% (Theobald, 2005).

While exurban residential households have the potential to further contribute to C storage by providing additions in the form of N fertilization and irrigation to the landscape, these additions have a small effect relative to implementing a strategy of no management. The advantages of these gains may be offset by a variety of factors. First, we showed that the removal of carbon from the system in the form of annual CWD and litter removals are substantially greater than the gains from our N fertilization and irrigation strategies. Second, we did not take into account the factors of production that create the fertilizer and additional water inputs that could contribute to CO_2 emissions and offset any gains through their deployment. Third, we assumed that C removed from the system was released to the atmosphere, when in many cases households will hide turfgrass litter in DTC or provide them to local municipalities that may sequester or recycle the organic contents.

These results provide us with a range of land-management actions and strategies to explore and evaluate their impacts on C storage and flux. Investigation of the types of policies that may alter the proportions of land cover, the placement of turfgrass litter, and the level of fertilization and irrigation have all been shown to affect C storage within a single residential parcel. The degree to which these land-management strategies can affect national carbon estimates and mitigate global climate change have not been fully explored. However, new data describing the proportions and patterns of land cover within residential parcels (Robinson, 2012) and large-scale studies that have estimated the total US turfgrass area to be greater than the area of any major food crop (Robbins and Birkenholtz, 2003) provide future avenues for this work to contribute to estimates of the impact of residential land policies on C storage.

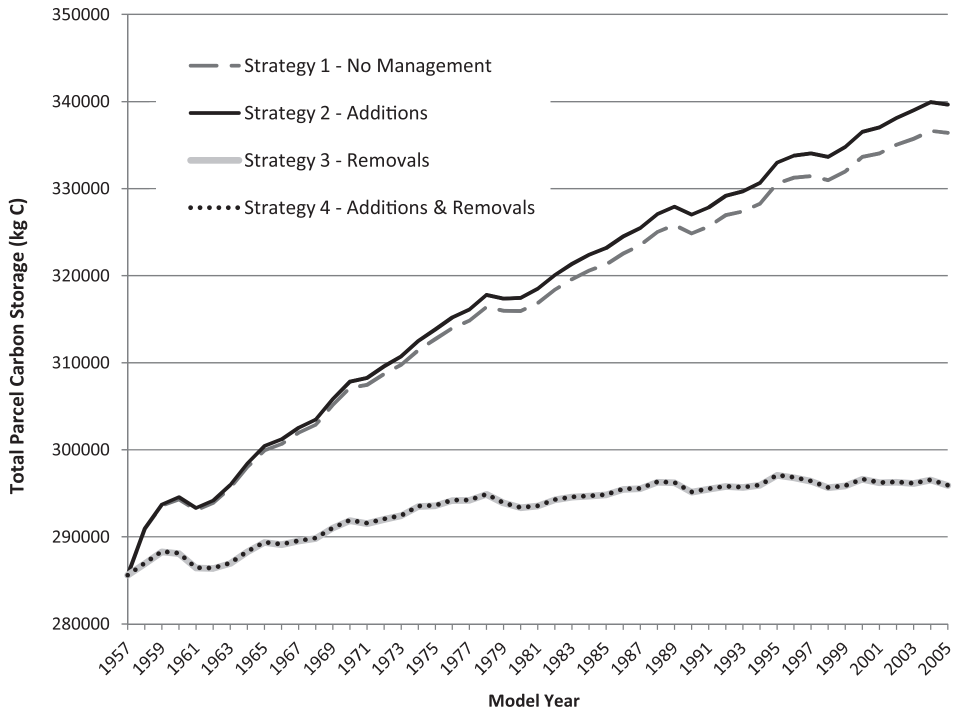

Fig. 7. Total parcel C storage by four different exurban residential land-management strategies.

5.2. Methodological issues associated with the integration of agent-based approaches and ecosystem models

We opted to integrate BIOME-BGC with our ABM of exurban development and land management for a variety of reasons. Its prior use across a variety of biomes including forests and grasslands through a change in parameter sets made it an attractive candidate to develop parameter sets for human-dominated residential land as a new conceptual biome. Its prior application to spatial scales from 30 m to heterogeneous regional landscapes was appropriate for our scale of analysis. Lastly, the combination of carbon, nitrogen, and water cycles provide a variety of entry points for which we can alter processes within those cycles based on land-management practices.

We chose to integrate BIOME-BGC with our ABM using a loose coupling approach to facilitate future linkage to a range of ecosystem process models. The benefits of loose coupling include: ease of integration, freedom from full understanding of the individual model components, and a framework that embraces independent model development and linking to other ecological models. The primary disadvantage of a loose-coupling approach is the computational overhead associated with maintaining starting conditions, modifying input files, and extracting results from output files.

In our application example, we ran BIOME-BGC once per year for 48 years with land management modifying input files (i.e. ecophysiology, meteorological, nutrient pools, and site characteristics) and output files (i.e. nutrient pools). The model was run at 16 different locations within each of four different residential parcels. The result is a possible 15,360 (5 files × 48 years × 16 locations × 4 parcels) file manipulations, which does not include the spinup required because we assumed soil and other site conditions were uniform across the four parcels. A single model run averaged about 3 min when run on different dual-core Linux machines. Assuming the same computational performance, a model run for a sample township at the urban-rural fringe, such as Scio Township, which is ~88 km^2 in size – a typical size for townships across the Midwest U.S. (Linklater, 2002) – with approximately 50% in turfgrass and DTC would take approximately 2312.5 min or 1.6 days. We are generous in this estimate as much of the area is likely not managed land and would therefore involve less interaction between the ABM and BIOME-BGC and lower processing times.

To lessen the overhead associated with file manipulation, CARVE can be used to aggregate and disaggregate land management and site conditions to a specified land unit. However, if CARVE aggregates heterogeneous land-management practices then the result is an averaging of vegetation growth response across the area aggregated. The degree to which this averaging affects individual and aggregate carbon storage across a parcel, subdivision, or region has not yet been investigated and will be the focus of future research.

Due to our loose coupling approach we had to make assumptions to represent different land-management practices as drivers of vegetation growth. One set of assumptions was associated with the timing of land-management practices. While BIOME-BGC runs at a daily time scale, the only daily variables read in by the model are the climate (e.g. precipitation, temperature, solar radiation) and atmospheric variables (CO_2 and N deposition). We represented irrigation at a daily time step by adding the irrigation quantities to precipitation. While land-management activities are more closely associated with daily than annual activities, access to the nutrient pools to implement litter and CWD removal (e.g. raking) or nutrient additions was only available at an annual time step.

We could potentially include two additional land-management approaches in future applications. The first is to represent actions like pesticide use by altering the ecophysiological parameters that represent variables like the probability of mortality. The second is to use the gap between potential values produced by BIOME-BGC and observed values to represent the possible gains due to management or technology improvements that cannot be explicitly incorporated via the ecosystem model parameters.

5.3. Future research directions

We acknowledge that many potential factors affecting C outcomes from residential land-management practices were excluded. Without a lifecycle analysis of the energy used for daily activities and management inputs, this research provides only a portion of the relative impact of different land-management strategies and behaviours on C storage. However, by adding management we have made extensive gains over previous research efforts that stop at estimates of the quantity and pattern of LULCC.

We identified a number of issues that require further attention. First, we represented patches of DTC but did not provide an avenue for representing individual trees or sparse woody vegetation intermixed with turfgrass, which is common on residential properties. The ecosystem model BIOME-BGC does not reduce to the resolution of a single tree that may also experience individual management actions such as fertilization. Other ecosystem models will need to be pursued in order to represent individual trees using the presented framework. However, application of this framework with models that operate on finer resolutions will increase data requirements and computational overhead, which may delay our ability to scale-up ABM results to improve our understanding of the broader impacts of land management at large spatial extents.

Second, many additional management activities may be implemented. Given the substantial role of DTC in residential C storage, future work could investigate the impacts of fertilization while the forest is young, how balanced fertilization and irrigation may be implemented, or how over fertilization may be avoided for both tree cover and turfgrass?

Third, edge effects have been shown to affect C storage (Robinson et al., 2009), but to what degree do they play a role among different patches of land cover on residential parcels? How can we deal with lateral transfers of energy and the relative impact of biophysical drivers such as site characteristics, landform, and climate relative to management practices?

6. Conclusions

Land-use and land-management dynamics jointly determine land-cover change and consequentially ecosystem function. This paper has presented a framework that extends traditional land-change models to incorporate land-market mechanisms, land-management behaviour, and by linking to an ecosystem process model (i.e. BIOME-BGC), estimates of ecosystem function. Each of these components contributes something novel to land-change science.

We have developed a flexible software framework that couples a LULCC model to a variety of ecosystem process models. By better representing the natural system in coupled natural-human land systems, we advance research called for by international and national institutions (e.g. Global Land Project, U.S. National Science Foundation's Coupled Natural/Human System program) and aim to address questions such as: what management strategies have unintended effects on carbon flux and storage with and without the presence of land markets; and how does residential demand for properties depend on current and projected land-management practices?

While contemporary land-change research has improved the coupling of natural and human system models (e.g. Yadav et al., 2008), we go one step further to include land-management practices. By improving our understanding of what drives landowners to perform specific land-management actions as well as how and to what degree land management can alter ecosystem function, we can understand the social, economic, and policy conditions that may lead a property to be a source or sink of carbon. By combining the processes of land-market exchange, land management, and ecosystem impact the framework allows us to ask policy questions that may help keep highly fragmented and human-dominated exurban landscapes acting as carbon sinks, essentially providing an ecosystem service that could help mitigate the effects of climate change.

Appendix A. Ecophysiology parameters for dense tree cover (deciduous)

Ecophys	DBF (deciduous broadleaf forest)	
1 (flag)	1 = Woody	0 = Non-woody
0 (flag)	1 = Evergreen	0 = Deciduous
1 (flag)	1 = C3 PSN	0 = C4 PSN
1 (flag)	1 = Model phenology	0 = User-specified phenology
0 (yday)	Yearday to start new growth (when phenology flag = 0)	
0 (yday)	Yearday to end litterfall (when phenology flag = 0)	
0.2 (prop.)	Transfer growth period as fraction of growing season	
0.2 (prop.)	Litterfall as fraction of growing season	
1.0 (1/yr)	Annual leaf and fine root turnover fraction	
0.70 (1/yr)	Annual live wood turnover fraction	
0.02 (1/yr)	Annual whole-plant mortality fraction	
0.00 (1/yr)	Annual fire mortality fraction	
1.2 (ratio)	(Allocation) new fine root C:new leaf C	
2.2 (ratio)	(Allocation) new stem C:new leaf C	
0.16 (ratio)	(Allocation) new live wood C:new total wood C	
0.22 (ratio)	(Allocation) new croot C:new stem C	
0.5 (prop.)	(Allocation) current growth proportion	
25.0 (kg C/kg N)	C:N of leaves	
55.0 (kg C/kg N)	C:N of leaf litter, after retranslocation	
48.0 (kg C/kg N)	C:N of fine roots	
48.0 (kg C/kg N)	C:N of live wood	
550.0 (kg C/kg N)	C:N of dead wood	
0.38 (DIM)	Leaf litter labile proportion	
0.44 (DIM)	Leaf litter cellulose proportion	
0.18 (DIM)	Leaf litter lignin proportion	
0.34 (DIM)	Fine root labile proportion	
0.44 (DIM)	Fine root cellulose proportion	
0.22 (DIM)	Fine root lignin proportion	
0.77 (DIM)	Dead wood cellulose proportion	
0.23 (DIM)	Dead wood lignin proportion	
0.045 (1/LAI/d)	Canopy water interception coefficient	
0.54 (DIM)	Canopy light extinction coefficient	
2.0 (DIM)	All-sided to projected leaf area ratio	
32.0 (m^2/kg C)	Canopy average specific leaf area (projected area basis)	
2.0 (DIM)	Ratio of shaded SLA:sunlit SLA	
0.07 (DIM)	Fraction of leaf N in Rubisco	
0.006 (m/s)	Maximum stomatal conductance (projected area basis)	
0.00006 (m/s)	Cuticular conductance (projected area basis)	
0.01 (m/s)	Boundary layer conductance (projected area basis)	
−0.34 (MPa)	Leaf water potential:start of conductance reduction	
−2.2 (MPa)	Leaf water potential:complete conductance reduction	
1100.0 (Pa)	Vapour pressure deficit:start of conductance reduction	
3600.0 (Pa)	Vapour pressure deficit:complete conductance reduction	

Appendix B. Ecophysiology parameters for turfgrass

Ecophys	C3 grass	
0 (flag)	1 = Woody	0 = Non-woody
0 (flag)	1 = Evergreen	0 = Deciduous
1 (flag)	1 = C3 PSN	0 = C4 PSN
0 (flag)	1 = Model phenology	0 = User-specified phenology
0 (yday)	Yearday to start new growth (when phenology flag = 0)	
364 (yday)	Yearday to end litterfall (when phenology flag = 0)	
1.0 (prop.)	Transfer growth period as fraction of growing season	
1.0 (prop.)	Litterfall as fraction of growing season	
1.0 (1/yr)	Annual leaf and fine root turnover fraction	
0.00 (1/yr)	Annual live wood turnover fraction	

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(*continued*)

Ecophys	C3 grass
0.0 (1/yr)	Annual whole-plant mortality fraction (herbivory)
0.0 (1/yr)	Annual fire mortality fraction
1.0 (ratio)	(Allocation) new fine root C:new leaf C
0.0 (ratio)	(Allocation) new stem C:new leaf C
0.0 (ratio)	(Allocation) new live wood C:new total wood C
0.0 (ratio)	(Allocation) new croot C:new stem C
0.5 (prop.)	(Allocation) current growth proportion
25.0[a] (kg C/kg N)	C:N of leaves
40.0 (kg C/kg N)	C:N of leaf litter, after retranslocation
50.0 (kg C/kg N)	C:N of fine roots
0.0 (kg C/kg N)	C:N of live wood
0.0 (kg C/kg N)	C:N of dead wood
0.68 (DIM)	Leaf litter labile proportion
0.23 (DIM)	Leaf litter cellulose proportion
0.09 (DIM)	Leaf litter lignin proportion
0.36 (DIM)	Fine root labile proportion
0.52 (DIM)	Fine root cellulose proportion
0.12 (DIM)	Fine root lignin proportion
0.75 (DIM)	Dead wood cellulose proportion
0.25 (DIM)	Dead wood lignin proportion
0.0225 (1/LAI/d)	Canopy water interception coefficient
0.48 (DIM)	Canopy light extinction coefficient
2.0 (DIM)	All-sided to projected leaf area ratio
70.0 (m^2/kg C)	Canopy average specific leaf area (projected area basis)
2.0 (DIM)	Ratio of shaded SLA:sunlit SLA
0.21 (DIM)	Fraction of leaf N in Rubisco
0.006 (m/s)	Maximum stomatal conductance (projected area basis)
0.00006 (m/s)	Cuticular conductance (projected area basis)
0.04 (m/s)	Boundary layer conductance (projected area basis)
−0.73 (MPa)	Leaf water potential:start of conductance reduction
−2.7 (MPa)	Leaf water potential:complete conductance reduction
1000.0 (Pa)	Vapour pressure deficit:start of conductance reduction
5000.0 (Pa)	Vapour pressure deficit:complete conductance reduction

[a] Value provided by the on-farm compostable handbook.

References

An, L., Linderman, M., Qi, J., Shortridge, A., Liu, J., 2005. Exploring complexity in a human- environment system: an agent-based spatial model for multidisciplinary and multiscale integration. Annals of Association of American Geographers 95 (1), 54–79.

Arthur, W.B., Durlauf, S.N., Lane, D., 1997. The Economy as an Evolving Complex System II. In: Santa Fe Institute Studies in the Science of Complexity, vol. XXVII. Addison-Wesley, Reading, MA, 608 pp.

Brown, D.G., Robinson, D.T., 2006. Effects of heterogeneity in residential preferences on an agent-based model of urban sprawl. Ecology and Society 11 (1), 46 [online] URL: http://www.ecologyandsociety.org/vol11/iss1/art46/.

Brown, D.G., Robinson, D.T., An, L., Nassauer, J.I., Zellner, M., Rand, W., Riolo, R., Page, S.E., Low, B., Wang, Z., 2008. Exurbia from the bottom-up: confronting empirical challenges to characterizing a complex system. Geoforum 39, 805–818.

Deadman, P.J., Robinson, D.T., Moran, E., Brondizio, E., 2004. Colonist household decisionmaking and land-use change in the Amazon Rainforest: an agent-based simulation. Environment and Planning B: Planning and Design 31, 693–709.

Fernandez, L.E., Brown, D.G., Marans, R.W., Nassauer, J.I., 2005. Characterizing location preferences in an exurban population: implications for agent based modeling. Environment and Planning B: Planning and Design 32 (6), 799–820.

Filatova, T., Parker, D., van der Veen, A., 2009a. Agent-based urban land markets: agent's pricing behavior, land prices and urban land use change. Journal of Artificial Societies and Social Simulation 12.

Filatova, T., van der Veen, A., Parker, D.C., 2009b. Land market interactions between heterogeneous agents in a heterogeneous landscape: tracing the macro-scale effects of individual trade-offs between environmental amenities and disamenities. Canadian Journal of Agricultural Economics 57 (4), 431–457.

Fontaine, C.M., Rounsevell, M.D.A., 2009. An agent-based approach to model future residential pressure on a regional landscape. Landscape Ecology 24, 1237–1254.

Gaube, V., Kaiser, C., Wildenberg, M., Adensam, H., Fleissner, P., Kobler, J., Lutz, J., Schaumberger, A., Schaumberger, J., Smetschka, B., Wolf, A., Richter, A., Haberl, H., 2009. Combining agent-based and stock-flow modelling approaches in a participative analysis of the integrated land system in Reichraming, Austria. Landscape Ecology 24, 1149–1165.

Howe, T.R., Collier, N.T., North, M.J., Parker, M.T., Vos, J.R., 2006. Containing agents: contexts, projections, and agents. In: Sallach, D., Macal, C.M., North, M.J. (Eds.), Agent 2006 Conference on Social Agents: Results and Prospects. Argonne National Laboratory and The University of Chicago.

Huigen, M.G.A., 2004. First principles of the MameLuke multi-actor modelling framework for land use change, illustrated with a Philippine case study. Journal of Environmental Management 72, 5–21.

Jung, M., LeMaire, G., Zaehle, S., Luyssaert, S., Vetter, M., Churkina, G., Ciais, P., Viovy, N., Reichstein, M., 2007. Assessing the ability of three land ecosystem models to simulate gross carbon uptake of forests from boreal to Mediterranean climate in Europe. Biogeosciences 4, 647–656.

Kok, K., Van Delden, H., 2009. Combining two approaches of integrated scenario development to combat desertification in the Guadalentín watershed, Spain. Environment and Planning B: Planning and Design 36, 49–66.

Linklater, A., 2002. Measuring America: How an Untamed Wilderness Shaped the United States and Fulfilled the Promise of Democracy. Walker Publishing Company, ISBN 0-8027-1396-3.

Milesi, C., Running, S.W., Elvidge, C.D., Dietz, J.B., Tuttle, B.T., Nemani, R.R., 2005. Mapping and modelling the biogeochemical cycling of turfgrasses in the United States. Environmental Management 36 (3), 426–438.

Monticino, M., Acevedo, M., Callicott, B., Cogdill, T., Lindquist, C., 2007. Coupled human and natural systems: a multi-agent-based approach. Environmental Modelling and Software 22, 656–663.

Parker, D.C., Brown, D.G., Filatova, T., Riolo, R., Robinson, D.T., Sun, S., 2012. Do land markets matter? A modeling ontology and experimental design to test the effects of land markets for an agent-based model of ex-urban residential land-use change. In: Batty, M., Heppenstall, A., Crooks, A. (Eds.), Spatial Agent-based Models: Principles, Concepts and Applications, pp. 525–542. http://dx.doi.org/10.1007/978-90-481-8927-4_26.

Rindfuss, R.R., Walsh, S.J., Turner II, B.L., Fox, J., Mishra, V., 2004. Developing a science of land change: challenges and methodological issues. Proceedings of the National Academy of Sciences of the United States of America 101, 13976–13981.

Robbins, P., Birkenholtz, T., 2003. Turfgrass revolution: measuring the expansion of the American lawn. Land Use Policy 30, 181–194.

Robinson, D.T., 2012. Land-cover fragmentation and configuration of ownership parcels in an exurban landscape. Urban Ecosystems 15, 53–69.

Robinson, D.T., Brown, D.G. , 2009. Evaluating the effects of land-use development policies on ex-urban forest cover: an integrated agent-based GIS approach. International Journal of Geographical Information Science 23 (9), 1211–1232.

Robinson, D.T., Brown, D.G., Currie, W.S., 2009. Modelling carbon storage in highly fragmented and human-dominated landscapes: linking land-cover patterns and ecosystem processes. Ecological Modelling 220, 1325–1338.

Running, S.W., Hunt, R.E., 1993. Generalization of a forest ecosystem process model for other biomes, BIOME-BGC, and an application for global-scale models. In: Ehleringer, J.R., Field, C.B. (Eds.), Scaling Physiological Processes: Leaf to Globe. Academic Press Inc., pp. 141–158.

Sarmiento, J.L., Sundquist, E.T., 1992. Revised budget for the oceanic uptake of anthropogenic carbon dioxide. Nature 356, 589–593.

Schaldach, R., Alcamo, J., Koch, J., Kölking, C., Lapola, D.M., Schüngel, J., 2011. An integrated approach to modelling land-use change on continental and global scales. Environmental Modelling and Software 26, 1041–1051.

Schreinemachers, P., Berger, T., 2011. An agent-based simulation model of human-environment interactions in agricultural systems. Environmental Modelling and Software 26, 845–859.

Schultz, W., 1999. A Man's Turf: the Perfect Lawn. Three River Press, New York.

Smajgl, A., Brown, D.G., Valbuena, D., Huigen, M.G.A., 2011. Empirical characterisation of agent behaviours in socio-ecological systems. Environmental Modelling and Software 26 (7), 837–844.

Sundquist, E.T., 1993. The global carbon dioxide budget. Science 259, 934–941.

Tatarinov, F.A., Cienciala, E., 2006. Application of BIOME-BGC model to managed forests. 1. Sensitivity analysis. Forestry and Ecological Management 237, 267–279.

Tesfatsion, L., Judd, K.L., 2006. Handbook of Computational Economics. In: Agent-Based Computational Economics, vol. II. Elsevier B.V..

Theobald, D.M., 2005. Landscape patterns of exurban growth in the USA from 1980 to 2020. Ecology and Society 10, 32.

Thornton, P.E., Law, B.E., Gholz, H.L., Clark, K.L., Falge, E., Ellsworth, D.S., Goldstein, A.H., Monson, R.K., Hollinger, D., Falk, M., Chen, J., Sparks, J.P., 2002. Modeling and measuring the effects of disturbance history and climate on carbon and water budgets in evergreen needleleaf forests. Agricultural and Forest Meteorology 113, 185–222.

Turner II, B.L., Lambin, E.F., Reenberg, A., 2007. The emergence of land change science for global environmental change and sustainability. Proceedings of the National Academy of Sciences of the United States of America 104, 20666–20671.

Verburg, P.H., Berkel, D.v., Doorn, A.v., Eupen, M.v., Heiligenberg, H.A.R.M.v., 2010. Trajectories of land use change in Europe: a model-based exploration of rural futures. Landscape Ecology 25 (2), 217–232.

White, A.M., Thornton, P.E., Runnin, S.W., Nemani, R.R., 2000. Parameterization and sensitivity analysis of the BIOME-BGC terrestrial ecosystem model: net primary production controls. Earth Interactions 4 (3), 1–85.

Yadav, V., Del Grosso, S.J., Parton, W.J., Malanson, G.P., 2008. Adding ecosystem function to agent-based land use models. Land Use Science 3, 27–40.

Zellner, M.L., Page, S.E., Rand, W., Brown, D.G., Robinson, D.T., Nassauer, J., Low, B., 2009. The emergence of zoning policy games in exurban jurisdictions: informing collective action theory. Land Use Policy 26, 356–367.