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Remote sensing the vulnerability of vegetation in natural terrestrial ecosystems

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ABSTRACT

Climate change is altering the species composition, structure, and function of vegetation in natural terrestrial ecosystems. These changes can also impact the essential ecosystem goods and services derived from these ecosystems. Following disturbances, remote-sensing datasets have been used to monitor the disturbance and describe antecedent conditions as a means of understanding vulnerability to change. To a lesser extent, they have also been used to predict when desired ecosystems are vulnerable to degradation or loss. In this paper, we review studies that have applied remote sensing imagery to characterize vegetation vulnerability in both retrospective and prospective modes. We first review vulnerability research in natural terrestrial ecosystems including temperate forests, tropical forests, boreal forests, semi-arid lands, coastal areas, and the arctic. We then evaluate whether remote sensing can evaluate vulnerability sufficiently in advance of future events in order to allow the implementation of mitigation strategies, or whether it can only describe antecedent conditions *a posteriori*. The majority of existing research has evaluated vulnerability retrospectively, but key studies highlight the considerable potential for the development of early warnings of future vulnerability. We conclude that future research needs to focus on the development of a greater number of remotely sensed metrics to be used in a prospective mode in assessing vulnerability of terrestrial vegetation under change.

1. Introduction

Climate change has modified disturbance regimes, altering the frequency, duration, and intensity of ecological disturbance processes (Bergeron & Archambault, 1993; Chapin et al., 2000; Flannigan, Stocks, & Wotton, 2000; Goetz, Bunn, Fiske, & Houghton, 2005; Westerling, Turner, Smithwick, Romme, & Ryan, 2011). Simultaneously, the ecological ranges of many tree species are changing as a function of climate change (Chmura et al., 2011; Crimmins, Dobrowski, Greenberg, Abatzogluo, & Mynseberge, 2011; Linder et al., 2010). These changes can push natural ecosystems outside their historic range of variability (Breshears et al., 2005; Landres, Morgan, & Swanson, 1999; Swetnam, Allen, & Betancourt, 1999), potentially resulting in inelastic regime shifts (Fig. 1, Beisner, Haydon, & Cuddington, 2003; Schefferm & Carpenter, 2003; Folke et al., 2004; McLauchlan et al., 2014). These changes in the distribution, structure, and function of terrestrial vegetation may result in a potential loss of desired natural capital in the form of specific ecosystem goods and services (Table 1), leading to wider social-economic and ecosystem impacts (Costanza & Daley, 1991; Costanza et al., 1997; de Groot, Wilson, & Boumans, 2002; Schröter et al., 2005). Disturbance regimes and species distributions are naturally dynamic (Carcaillet et al., 2001; Davis & Shaw, 2001; Nowak, Nowak, Tausch, & Wigand, 1994; Whitlock, Shafer, & Marlon, 2003), providing challenges in understanding the vulnerability of the associated ecosystem goods and services to climate change (Schröter et al., 2005). Considerable interest has focused on identifying if, where, and when the ecosystem goods and services are impacted by degradation or loss of the relevant terrestrial ecosystem (NRC, 2010). Given the large spatial scales over which terrestrial vegetation is evaluated and monitored, remote sensing is a logical tool to evaluate their vulnerability. Changes that are too subtle to notice at the local level may be significant when summarized at the synoptic scales captured by remote sensing data. However, given most ecosystem goods and services are not directly measurable by remote sensing datasets (Table 1), a challenge for the remote sensing community is to

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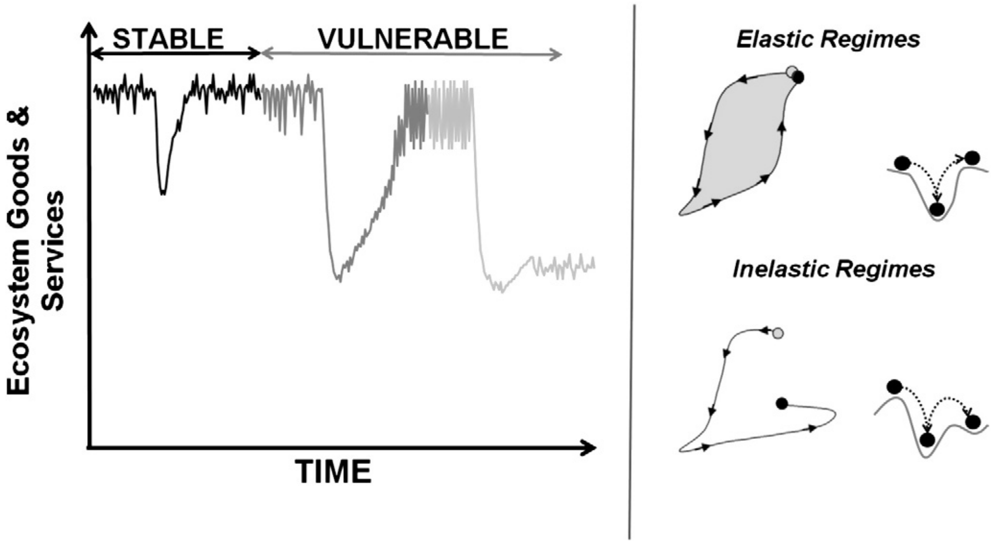

Fig. 1. Under stable conditions, a disturbance will perturb the system but will elastically return the system to pre-perturbation conditions. In a moderately vulnerable condition, increased variability may occur. Although disturbances may cause more pronounced impacts, the system will likely elastically return to pre-perturbed conditions; likely over longer time intervals and with increased variability. A highly vulnerable system has reached an ecological "tipping point" where a perturbation produces an inelastic change, leading to a new regime and potentially a complete loss in the original ecosystem service. Mitigation and adaptation strategies can lead to alternate regimes on a gradient, where the original (to a lesser degree) or alternate ecosystem goods and services may be attainable. The elasticity-hysteresis concepts are adapted from Schefferm and Carpenter (2003) and Folke et al. (2004).

identify quantitative metrics that can mechanistically bridge between the observed climate change impacts on natural terrestrial vegetation and the associated ecosystem goods and services (Fig. 2, Table 2).

Vulnerability indicators are often developed in order to describe the degree to which a system is susceptible to being impacted by future change (Alessa, Kliskey, & Brown, 2008; Alessa, Kliskey, Lammers, et al., 2008; Cutter, 1996; Hinkel, 2011; Ionescu, Klein, Hinkel, Kavi Kumar, & Klein, 2009; Timmerman, 1981; Villa & McLeod, 2002). Although hundreds of case-specific definitions of vulnerability have been created (Hinkel, 2011; Janssen & Ostrom, 2006; Linder et al., 2010), a review of these formulations is beyond the scope of this study. For the purposes of this review, we simply define vulnerability indicators as any quantitative metric using active or passive remotely sensed data that can be used to infer a "*probabilistic measure of possible future harm*" (Hinkel, 2011; Turner et al., 2003). For example, commonly accepted definitions of harm could include species mortality and economic loss; where a remote sensing analogue could be decreased primary productivity of a critical tree species or crop. In terms of climate change research, vulnerability indicators are intended to describe the susceptibility of the system to climate variability and extremes

Table 1
Common ecosystem goods and services.

Category	Example Ecosystem Goods and Services
Water supply, availability, and filtration	Quantity and quality of fresh water for reservoirs, irrigation, and industry [1, 2, 6, 7, 8, 9]
	Storage of water reserves in reservoirs, aquifers, and watersheds [1,2, 5, 7, 9]
	Drainage and irrigation [1,2]
	Water purification [4, 7]
	Snowpack depth, coverage, and ablation rates [3]
Erosion control	Soil retention by vegetation on steep slopes, reduction of erosion removal processes by wind and water [1,5, 7]
Waste processing	Removal and reduction of pollutants in watersheds [1, 5, 7, 9]
	Reduction of noise, dust, and fire pollutants in airsheds [2]
	Decomposition and detoxification [4]
Soils	Maintaining soil properties for agriculture, forestry, and recreation [2, 8]
Vegetation biodiversity. productivity, and reproduction: timber supply, food supply, and bioenergy supply	Quantity and quality of timber for paper, specialty wood products, and lumber [1,2, 7]
	Agricultural livestock and crop yields, fodder, fisheries, and bee hives [1,2, 5, 7]
	Wild terrestrial foods: game animals, wild fisheries [7]
	Seed dispersal and pollination of wildflowers and crops [1, 2, 4, 7, 9]
	Disease and pest regulation [7]
	Production of bioenergy crops, fiber, and fuels [3, 7]
Nutrient and biogeochemical cycling	Photosynthesis, decomposition, and nitrogen fixation [1, 6]
	Carbon sequestration and storage [3, 8]
Genetic resources	Harvest of plants for medicinal purposes [1,2, 7, 9]
Gas exchange	Maintenance of good air quality [2]
Disturbance dampening	The composition and structure of the vegetation (e.g., coral reefs, wetlands) can act to reduce the impacts of storms, floods, and droughts [1, 5, 7, 8]
	Establishment of refugia and seed banks [2]
Recreation and cultural sites	Access to non-commercial recreation and cultural sites such as State and National Parks, National Monuments. [1,2, 5]
	Sustainability of fishing, swimming, hiking, and skiing access areas [1,2,3, 9]
	Aesthetic, spiritual, and religious sites [7]
	Sites of scientific and educational interest [1,2]

[1]Costanza et al. (1997), [2]de Groot et al. (2002), [3]Schröter et al. (2005), [4] Kremen (2005), [5] Costanza and Daley (1991)
[6] Kessler, Salwasser, Cartwright, and Caplan (1992), [7] Carpenter et al. (2009), [8] Nelson et al. (2009), [9] Boyd and Spencer (2007)

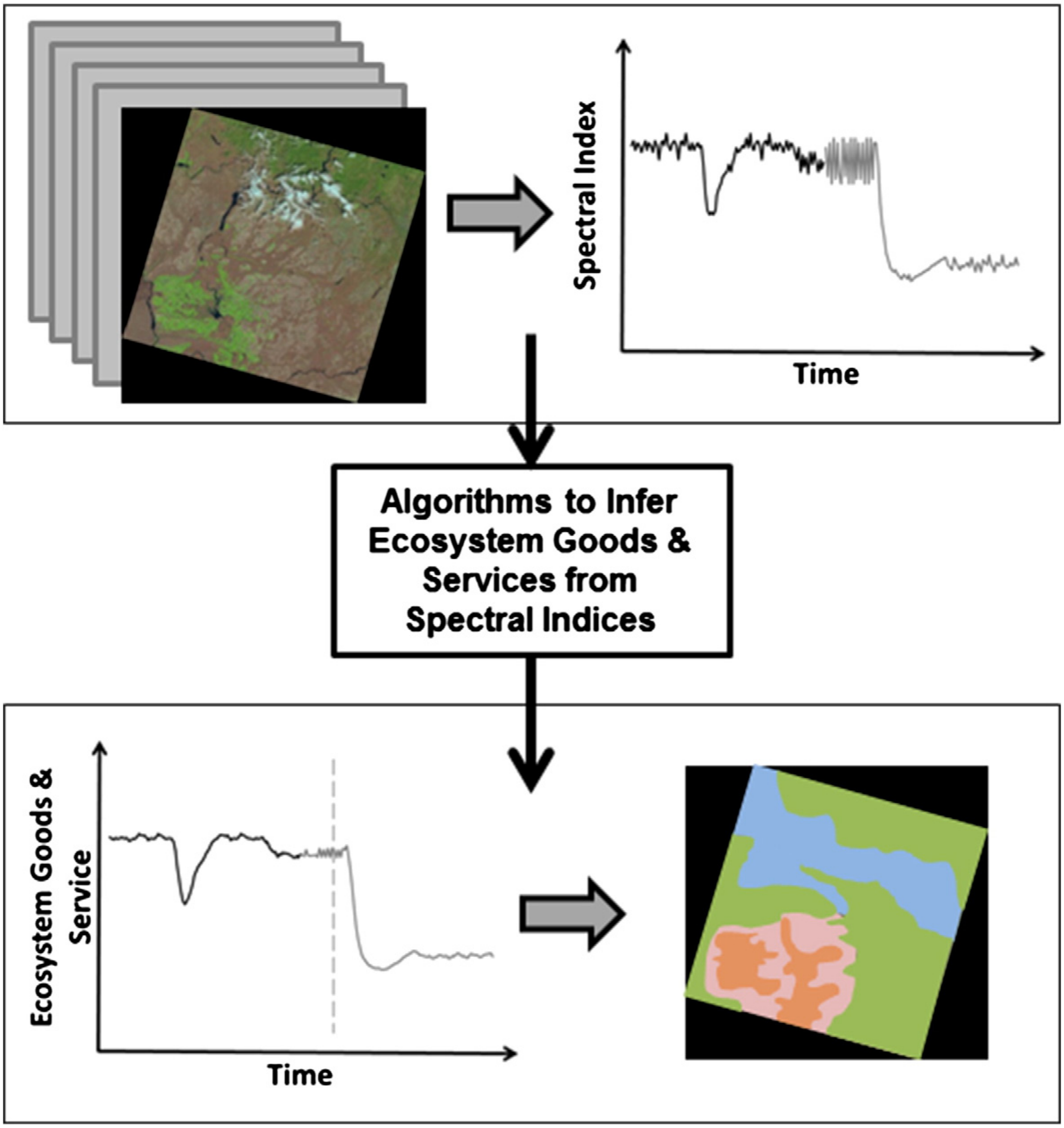

Fig. 2. Evaluating ecosystem goods and services requires the development of algorithms to connect between (i) remotely sensed metrics of vegetation that are sensitive to the changes in climate and (ii) the desired ecosystem goods and services (e.g., regressions between spectral indices related to water stress and crop yield). The upper insert shows Landsat 7 imagery of agricultural landscape in Washington State, USA (acquired 8/6/2013, Path 44, Row 27). The classified lower right image represents a hypothetical vulnerability map that could be associated with the ecosystem just prior to a tipping point.

(Casalegno, Amatulli, Camia, Nelson, & Pekkarinen, 2010; McCarthy, Canziani, Leary, Dokken, & White, 2001; Metzger, Schroter, Leemans, & Cramer, 2008). Vulnerability is often defined through the terms exposure, sensitivity, and resilience (Romer et al., 2012; Turner et al., 2003); although resilience is also often interchanged with adaptive capacity (Hinkel, 2011; Linder et al., 2010; McCarthy et al., 2001). *Exposure* generally describes how many of the components within the system are at risk and how long those components are in contact with the stressor (Kasperson & Kasperson, 2001; Kasperson et al., 2005; Romer et al., 2012; Sonwa, Somorin, Jum, Bele, & Nkem, 2012; Taubenbock et al., 2008), *sensitivity* describes the magnitude of the stressing event that the system will resist or absorb without significant change (Holling, 1973; Klaus, Holsten, Hostert, & Kropp, 2011; O'Brien, Leichenko, et al., 2004; O'Brien, Syhna, & Haugen, 2004 and Romer et al., 2012), and *resilience* describes the magnitude that the ecosystem and its components can resist, adjust, or absorb impacts from the stressor without the system being pushed into a different state (Folke et al., 2004; Holling, 1973; Klaus et al., 2011; O'Brien, Leichenko, et al., 2004; O'Brien, Syhna, et al., 2004; Romer et al., 2012; Taubenbock et al., 2008).

Remote sensing has been widely used to evaluate vegetation trajectories following disturbances in various ecosystems (Goetz, Mack, Gurney, Randerson, & Houghton, 2007; Hicke et al., 2003; Huang et al., 2010a; Kennedy, Cohen, & Schroeder, 2007; Kennedy, Yang, & Cohen, 2010). Some vulnerability studies have used such disturbance recovery trajectories *a posteriori* to determine that the system was vulnerable before the disturbance (Duguy et al., 2012; Nepstad et al., 2004), because the system responded adversely, it therefore must have been vulnerable. Whether or not ecological disturbances are positive or negative is a value judgment made by humans. “Severe” or “extreme” disturbances imply a fundamental shift in ecosystem condition, usually to what's deemed a less desirable state because of diminished ecological goods or services (Lannom et al., 2014; Lentile et al., 2006). Although vegetation composition, structure, and function may change in response to climate change, the ability to derive certain ecosystem goods and

Table 2
Examples and proposed remote sensing vulnerability metrics.

Metric	Potential usage
Productivity (species, forest) [1]	- Define a productivity lower limit (i.e. minimum sustainable economic threshold)
Composition [2–4]	- Presence/ability of vegetation to reestablish following disturbance or in response to external changes
Structure [5–7]	- Define canopy cover (CC) lower and upper limits CC% < ok < CC% where ecosystem function is altered.
Phenology [8–16]	- Define seasonality factors (growing season start, duration) that limit the ability to obtain a desired ecosystem service. - Detecting changes in phenology (leaf out, leaf duration) that are outside the historic range of variability.
Moisture [17–18]	- Define a moisture content threshold where vegetation would be susceptible a myriad of disturbances (fire, drought induced die-off, hard freeze events, etc.,) - Define soil moisture content thresholds were microbial greenhouse gas emissions become sinks or sources.
Response to Prior Disturbances and Changes [19–20]	- Evaluate the drivers, patterns, and processes of recovery following past disturbances and other drivers of global change.
Nutrient Cycling [21–22]	- Define canopy nitrogen thresholds at which ecosystem goods and services are sustained - Define nitrogen deposition thresholds for vegetation composition, structure, and function.
Meteorology and climate [23]	- Define thresholds in micro-climate and radiative transfer data to determine when species shifts are likely.
Topography [24]	- Define locations where vegetation composition, function, and structure will be limited under future scenarios (urban growth, landslides, flooding, sea-level rise).
Water Availability and Salinity [25–26]	- Define thresholds (upper and lower bounds) where productivity is maintained and not inhibited by too little or too much available water
Elevation of snow-rain transition, Snow ablation dates and rates [27–28]	- Define the landscape thresholds for the snow-rain transition needed to sustain an ecosystem service. - Define thresholds of seasonality where productivity is maintained
Snow volume and snow water equivalent [29]	- Define thresholds of snow pack volume and snow water equivalent (SWE) necessary to sustain the desired ecosystem service

[1] Lobell et al. (2002), [2] Coops and Waring (2011), [3] Waring et al. (2011), [4] Coops et al. (2012), [5] Fiala, Garman, and Gray (2006), [6] Chopping et al. (2008), [7] Smith et al. (2009), [8] Reed et al. (1994), [9] Goetz and Prince (1996), [10] Hicke et al. (2003), [11] Sims and Gamon (2002), [12] Gamon et al. (1997), [13] Zhang et al. (2003), [14] Kimball et al., (2004), [15] Reed (2006), [16] White et al. (2009), [17] Njoku and Entekhabu (1996), [18] Jackson et al. (2003), [19] Hicke et al. (2003), [20] Goetz et al. (2007), [21] Martin and Aber (1997), [22] Hansen and Schjoerring (2003), [23] Crimmins et al. (2011), [24] Cooper et al. (2013), [25] Bourgeau-Chavez et al. (2005), [26] Zinnert et al. (2012), [27] Dozier et al. (2009), [28] Nolin (2010), [29] Tinkham et al. (2014)

services may change at different rates and might only be observable following extreme events such as disturbances or regime shifts (Fig. 1). To evaluate whether certain ecosystem goods and services may become susceptible to degradation or loss under future climate scenarios, what is needed are vulnerability indicators that can provide *a priori* warnings of future reductions in ecosystem goods and services. This real-time early detection could enable remote sensing datasets to provide a considerable advantage in assessing ecosystem goods and services under projected climate change. As such, compelling questions that arise are:

(1) Can remote sensing datasets be used to evaluate vulnerability of ecosystems traits such as species composition, structure, and function?
(2) Can remote sensing datasets be used to effectively evaluate vulnerability of critical ecosystem goods and services?
(3) Can we detect vulnerability of ecosystem goods and services before a disturbance occurs? Or conversely, can remote sensing only be used to evaluate antecedent conditions and recovery trajectories after a disturbance has occurred?

In terms of (1), several synthesis studies and special issues have highlighted the potential and limitations of remote sensing to characterize plant function types, composition, and structure (Smith, Greenberg, & Vierling, 2008; Smith, Wynne, & Coops, 2008; Ustin & Gamon, 2010). In contrast, this paper is focused only on studies that sought to use such metrics as part of a vulnerability assessment. To achieve this review we principally evaluated studies that included the word vulnerability in addition to phrases implying analysis with geospatial datasets. We also included studies focused on related topics such as the ability of remote sensing to assess vegetation stress due to water, nutrients, and heat (Haung & Anderegg, 2012; Hilker, Coops, Wulder, Black, & Guy, 2011; Michaelian, Hogg, Hall, & Arsenault, 2011), the assessment of post-disturbance recovery (Goetz et al., 2007; Hicke et al., 2003), and studies seeking to quantify ecosystem goods and services (Costanza & Daley, 1991; Costanza et al., 1997; de Groot et al., 2002; Schröter et al., 2005). To provide context, this review includes a brief section highlighting expected changes in terrestrial vegetation and related process under projected climate change. The review is then organized around natural terrestrial vegetation ecosystems highlighted as being particularly vulnerable to climate change, including temperate forests, tropical forests, boreal forests, semi-arid lands, coastal areas, and the arctic. Managed ecosystems such as agricultural systems and plantations are not included in this review. Finally, we discuss the application of remotely sensed data to construct ecologically-based modeling and decision support tools to enable improved forecasting of when and where future vulnerabilities are likely to occur.

2. Background: projected change in terrestrial vegetation under climate change

Our understanding of how climate change will impact natural terrestrial ecosystems and the goods and services they provide comes primarily from four sources: paleo-ecological research, recent observations of change in response to variation in climate, controlled experiments, and model simulations based on physiological processes. Among these information sources, there is consensus that changes in temperature and precipitation regimes will affect the composition, structure, and function of terrestrial ecosystems (Carcaillet & Brun, 2000; Chapin et al., 2000; Crimmins et al., 2011; Davis & Shaw, 2001; Nowak et al., 1994; Parmesan, 2006; Rehfeldt & Jaquish, 2010; Wu, Dijkstra, Koch, Peñuelas, & Hungate, 2011). Changes in terrestrial vegetation in response to climate have begun to be observed within remote sensing datasets, particularly within more climatically extreme regions. Studies based in the Alaskan Arctic have found longer growing seasons and thawing permafrost have led to increased productivity, with multitemporal imagery being used to monitor the expansion of shrubs into tussock tundra over the last fifty years (Kittel, Barker, Higgins, & Haney, 2011; Sturm, Racine, & Tape, 2001). At the other end of the temperature spectrum, the Normalized Difference Vegetation Index (NDVI: Rouse, Haas, Schell, & Deering, 1974) has been used to evaluate the response of vegetation within the African Sahel to increasing precipitation (Herrmann, Anyamba, & Tucker, 2005). The temperature and precipitation regimes under which tree species are currently able to establish and survive have been well described in the northwestern United States (Chmura et al., 2011; Rehfeldt, Ferguson, & Crookston, 2008), Europe (Linder et al., 2010), and tropical ecosystems (Laurance et al., 2011). These regimes have been used to predict how the distributions of individual tree species will be influenced by future climate using both correlative (Rehfeldt, Crookston, Saenz-Romero, & Campbell, 2012) and quasi-mechanistic approaches (Coops, Wulder, & Waring, 2012). Several recent reviews have highlighted how anthropogenic changes in climate are likely to affect the biogeography and structure of different ecosystems, including Australian ecosystems (Laurance et al., 2011), European forests (Linder et al., 2010), North American forests (Chmura et al., 2011 and Rehfeldt et al., 2012), boreal and arctic systems (Chapin et al., 2000), semi-arid and arid riparian zones (Perry, Andersen, Reynolds, Nelson, & Shafroth, 2012), and coastal ecosystems (Gilman, Ellison, Duke, & Field, 2008 and Simas, Nunes, & Ferreira, 2001). Specifically, a recent expert panel identified ten Australian ecosystems that they judged to be the most vulnerable to exhibiting large changes in ecosystem structure, condition, and function in response to small environmental changes (Laurance et al., 2011). These ecosystems included: mountainous environments, temperate eucalypt forests, tropical forests and savannahs, Mediterranean and semi-arid lands, floodplains and wetlands, and coral reefs (Laurance et al., 2011). In Europe, Schröter et al. (2005) highlighted that Mediterranean and mountainous ecosystems were particularly vulnerable. This section does not seek to repeat such reviews, but rather will briefly highlight expected changes and associated metrics that could be obtained from remote sensing datasets.

Projected shifts in ecosystem condition, structure, and function will likely manifest differently, even across small spatial extents, and will strongly influence both energy and water balances (Chmura et al., 2011; Easterling & Apps, 2005). With projected increases in the northwestern United States mean temperatures of 0.8–2.9 °C by 2050 and 1.6–5.4 °C by 2100, ecophysiological responses will likely result in changes in ecosystem composition (Chmura et al., 2011). Although species migration of flora has been projected through future climate space, the underlying geology and terrain will also influence species establishment and productivity (Barnett, Lambert, & Fry, 2008; Chmura et al., 2011) and often existing and potential disturbance regimes are ignored (Abatzoglou & Kolden, 2011). Geological drivers manifest especially through their influence on soil texture and depth (White, Scott, Hirsch, & Running, 2006), which affect the retention of soil moisture available for root uptake. Although plant responses will be ecosystem dependent, remote sensing will be able to provide broad-scale assessments of productivity, growing season length, and changes in precipitation timing and intensity (Chmura et al., 2011; Leung et al., 2004; Wu et al., 2011). An example of broad spatial-scale assessments of precipitation that can be obtained from remote sensing is data from NASA's Tropical Rainfall Monitoring Mission (TRMM) (Mahli et al., 2009). Similarly, soil moisture data has been widely available from microwave datasets, such as from the Advanced Microwave Scanning Radiometer (AMSR-E) sensor (Jackson, Lakshmi, Chan, & Nghiem, 2003; Njoku & Entekhabu, 1996) and the Soil Moisture Active Passive (SMAP) mission (Piles, Entekhabi, & Camps, 2009). Expected changes in precipitation due to increased winter temperatures include the phase of precipitation (i.e. rain instead of snow; Barnett, Pierce, et al., 2008; Leung et al., 2004), which may alter regional water budgets. A predicted impact of increased temperatures is the reduction of snow-pack due to increases in the elevation of the snow-rain transition zone in many mountainous ecosystems, leading to reduced spring and summer water supply in connected riparian ecosystems (Diffenbaugh & Field, 2013; Elsner et al., 2010; Schröter et al., 2005). Considerable research has demonstrated approaches to characterize the elevation, spatial extents, and volumes of mountainous snow using both passive and active remote sensing (Dozier, Green, Nolin, & Painter, 2009; Nolin, 2010; Tinkham, Smith, Marshal, Link, & Falkowski, 2014).

In terms of impacts on ecosystems, anthropogenic-driven climate change is expected to manifest as increases in atmospheric CO_2 that may lead to increases in productivity in areas not limited by water or nutrient availability (Easterling & Apps, 2005; McGuire, Chapin, Walsh, & Wirth, 2006; Phillips, Lewis, Baker, Chao, & Higuchi, 2008) and higher mortality and turnover rates in some ecosystems (Mahli et al., 2009). Increases in temperature are projected to increase ecosystem photosynthesis and respiration rates (Mahli et al., 2009; Wu et al., 2011), alter growing season timing and lengths (Abatzoglou & Kolden, 2011; Morissette et al., 2009), and increase plant water stress via increasing leaf-to-air vapor pressure deficits (Mahli et al., 2009). However, elevated CO_2 will likely increase water use efficiency (Marshall & Linder, 2013) and thus might partially offset water stress associated with increased temperatures (Mahli et al., 2009). Other documented impacts include: (i) increased variance in inter-annual precipitation, which could result in increased late summer moisture deficits and lower fuel moisture contents during fire seasons (Abatzoglou & Kolden, 2011; Littell et al., 2010); (ii) projected rises in sea-level impacting the depth at which soils are available to vegetation in coastal ecosystems such as coastal wetlands and mangrove forests (Cooper, Chen, Fletcher, & Barber, 2013; Gilman et al., 2008; Kasischke et al., 2010; Laurance et al., 2011; Simas et al., 2001); and (iii) changes in ecosystem structure due to species shifts, invasions, and ecosystem conversions (i.e. forest to savanna, tundra to forest) (Abatzoglou & Kolden, 2011; McGuire et al., 2006). Specifically, in agricultural systems crops will likely initially be more productive under increased CO_2 but it is expected that this will be partially offset when temperatures increase beyond ideal photosynthetic conditions or when water resources becomes a limiting factor (Easterling & Apps, 2005). In boreal forests high temperatures through sustained drought have led to observed decreases in net primary productivity (Beck et al., 2011; Sellers et al., 1997), potentially mitigated by longer growing seasons (Beck et al., 2011; Easterling & Apps, 2005). These and other expected climatic changes will result in direct impacts on vegetation composition, structure, and function. There will also be coupled system changes in terms of insect, disease, and fire regimes (Boisvenue & Running, 2006; Easterling & Apps, 2005; Hansen et al., 2001; Littell et al., 2010; Mohan, Cox, & Iverson, 2009).

Although our understanding of the impact of climate change on ecosystems is not unsophisticated, it is still difficult to project how climate change will impact specific ecosystems and the resultant ecosystem goods and services (Diffenbaugh & Field, 2013; Montoya & Raffaelli, 2010; Schröter et al., 2005; Tylianakis, Didham, Bascompte, & Wardle, 2008; Walther, 2010). In part, this is because individual species will respond to different climate signals and at different rates (Walther, 2010). Although climate change may limit the establishment of certain species (Coops & Waring, 2011; Coops et al., 2012), the impacts on certain ecosystem goods and services (e.g., timber yield) might not be observable until following a disturbance or regime shift. This uncertainty is particularly apparent in regards to terrestrial productivity and carbon cycling (Luo, 2007). Although photosynthesis and respiration are sensitive to temperature shifts, results from field experiments suggest that changes in community composition, nutrient cycling, water availability, and growing season length may ultimately control carbon dynamics at the ecosystem scale (Luo, 2007). A meta-analysis of climate change experiments found a generally positive response of net primary productivity to warming (Wu et al., 2011), but the number of experiments was limited and others have pointed out the highly variable response to experimental warming among plant species (Luo, 2007). Likewise, recent climate change appears to have generally increased forest productivity since the beginning of the 20th century, although there is considerably less certainty in regional forest productivity trends (Boisvenue & Running, 2006). Further, many ecosystems are expected to experience an increase in fire frequency and intensity in response to climate change, potentially causing or accelerating changes in ecosystem structure and composition (Chapin et al., 2000; Davidson, Williamson, & Parkins, 2003; Duguy et al., 2012; Flannigan et al., 2000; Goetz et al., 2005; Littell et al., 2010; Nepstad et al., 1999; Podur, Martell, & Knight, 2002). Increases in temperature have also led to well documented expansions of species like mountain pine beetle into the boreal forests of North America (Safranyik et al., 2010), which have been posited to increase fire potential and cause changes in species composition. Research in tropical forests has highlighted feedbacks between climate, fire, and logging that could lead to widespread forest loss (Cochrane, 2003; Mahli et al., 2009; Nepstad et al., 1999). The utility of remote sensing data to observe these changes will be most effective following disturbances within systems that are more sensitive to climate regime shifts (Coops et al., 2012; Linder et al., 2010; Rehfeldt et al., 2012).

3. Remote sensing of vulnerability in temperate forests

Studies assessing the impacts of climate change on temperate ecosystems have highlighted the occurrence of forest die-off events (Adams et al., 2012; Allen et al., 2010) and the likely lowering of productivity due to extended droughts and elevated temperatures (Diffenbaugh & Field, 2013). Temperate forests are vulnerable to the effects of anthropogenic climate change and land use on wildfire regimes, and remote sensing has particular utility for measuring, monitoring, and developing indicators for wildfire-related impacts on vegetation (Diaz-Delgado, Llorett, & Pons, 2003; Hardy, 2005; Lentile et al., 2006; Smith, Eitel, & Hudak, 2010; Smith, Lentile, Hudak, & Morgan, 2007; Smith et al., 2005). There are numerous short-term fire danger assessment products in North America, Europe, and Australia that transform remotely sensed data into a metric of fire potential based on vegetation stress (Burgan, Hartford, & Eidenshink, 1996; Chuvieco et al., 2004; Paltridge & Barber, 1988). There are equally a large number of studies that have evaluated post-fire trajectories in vegetation structure and condition using spectral remote sensing (Diaz-Delgado et al., 2003; Goetz et al., 2007; Hicke et al., 2003; Lannom et al., 2014; Lentile et al., 2006, 2009; Smith et al., 2007). Although these are retroactive assessments, evaluating whether ecosystems respond differently to different disturbances (e.g., fire, insects, and hurricanes) over decades as a function of climate change may provide useful information to evaluate whether a system is within a stable or vulnerable state (Fig. 1).

The majority of these vulnerability assessments are conducted *a posteriori* as in many cases the disturbances (e.g., wildfire, hurricane) had to occur prior to research being conducted. The need for *a priori* vulnerability assessments has been acknowledged (Duguy et al., 2012; Rehfeldt, Ferguson, & Crookston, 2009), although few have been applied. Common approaches are to apply and validate methods over past time periods (Duguy et al., 2012) or to model simulations of future climatic regimes (Casalegno et al., 2010; Coops & Waring, 2011; Waring, Coops, & Running, 2011). Studies have conducted *a posteriori* assessments of aspen (*Populus tremuloides Michx.*) die-back following extended droughts through a combination of basic satellite sensor classifications and aerial photograph surveys to assess extent of die-back as a function of climatic drought variables (Michaelian et al., 2011). Haung and Anderegg (2012) highlighted the utility of remote sensing data to systematically monitor the condition of aspen over large spatial extents via an example to quantify the loss in aboveground aspen biomass through linear spectral unmixing of Landsat imagery. This study however noted that such assessments likely exhibit considerable bias due to a lack of 3-dimensional vegetation structure data (Haung & Anderegg, 2012). In response to concern regarding the die-back of aspen and other large changes in species composition, several large monitoring programs that incorporate field measurements and remote sensing data have been initiated such as The Canadian *Climate Impacts on Productivity and Health of Aspen* and the United States National Park Service's *Vital Signs* programs (Fancy, Gross, & Cart, 2009; Michaelian et al., 2011).

In a pine forest and Mediterranean shrub sites, Duguy et al. (2012) conducted *a posteriori* vulnerability assessment of soil properties immediately following a fire and combined this with site information on typical fire return intervals; the resultant predictions of vulnerability were validated against NDVI trajectories following historical fires. To develop *a priori* metrics that are broadly transferable across different ecosystems it is necessary to characterize and apply approaches that preserve mechanistic connections between remote sensing metrics and surface properties. Indeed, several studies have highlighted the potential of using remote sensing datasets that incorporate vegetation dynamics, condition, and function for future vulnerability assessments (Duguy et al., 2012). In an assessment of vulnerability to insect damage in balsam fir (*Abies balsamea*) forests, Luther, Franklin, Hudak, and Meades (1997) combined field measures of leaf area index (LAI), forest inventory plots, basal area increment (BAI) data, a measure of vigor (BAI/sapwood basal area), and Landsat Thematic Mapper surface reflectance within a logistic regression to predict future vulnerability of these forests to both insect outbreaks and potential defoliation following an outbreak. Here, vulnerability was defined as a rating proportional to the amount of defoliation and the likelihood that a tree would recover from insect damage. However, the authors concluded that improved assessments would need better remote sensing data of vertical structure, productivity, and area of defoliation (Luther et al., 1997).

A series of studies that further extended the mechanistic capabilities of remotely sensed data to assess *a priori* ecosystem vulnerabilities compared modeled physiological attributes of tree species to projected changes in climate (Coops & Waring, 2011; Waring et al., 2011). The basis of this vulnerability metric, which is readily transferable to other ecosystems, was the suitability of a given species to the future climate regimes. Vulnerable locations were defined by a majority of years being unsuitable for that species, whereas resilient locations were defined by the inverse scenario; a majority of years favorable for growth of that species (Coops & Waring, 2011; Coops et al., 2012; Waring et al., 2011). The premise of this approach is that species that are resilient to climate change will continue to establish, while vulnerable species will be removed from the system (Coops & Waring, 2011; Coops et al., 2012; Nitschke & Innes, 2008). The outputs of this approach included LAI, length of growing season, and likelihood of disturbance; each of these is observable by remote sensing methods (Waring et al., 2011). In a related manner, Gritti, Smith, and Sykes (2006) assessed the vulnerability of Mediterranean vegetation to invasive species via application of LPJ-GUESS (a generalized ecosystem model) to predict LAI under future climate and disturbance scenarios. Although not a remote sensing study, Lexer et al. (2002) described a climate change index for Austrian forests that outlined degrees of vulnerability based on tree mortality. Lexer and Seidl (2009) further presented several forestry vulnerability indicators that were relevant to productivity, timber and carbon stocks, biodiversity, and disturbances. The highest weights were applied to the metrics that described changes in gross stem wood productivity, changes in average timber stock, and changes in salvage quantity relative to gross productivity; the latter incorporated the impacts of disturbances (Lexer & Seidl, 2009). These studies highlight the mechanistic potential of remote sensing datasets, and promote the use of other sensors to validate projections.

4. Remote sensing of vulnerability in tropical forests

Tropical forests are widely monitored for their vulnerability to global change (particularly the Amazon Basin). There has also been considerable research on using remote sensing data to provide spatially explicit inputs to ecosystem state-and-transition models for this biome that identify processes and feedbacks between deforestation, sub-canopy drying, and fire (Cochrane, 2003; Cochrane & Schulze, 1999; Cochrane et al., 1999; Hirota, Holmgren, Van Nes, & Scheffer, 2011). Several vulnerability studies have built upon agreement in the literature that intact tropical forests with closed canopies are more resilient to projected periods of extended drought under climate change as compared to open canopy forests, due to positive feedbacks that arise between deforestation and increased susceptibility to vegetation mortality from fire (Cochrane, 2003; Diffenbaugh & Field, 2013; Laurance & Williamson, 2001; Mahli et al., 2008, 2009; Nepstad et al., 1999). Similar fire-climate feedbacks, which are sometimes non-linear, have been detailed in equatorial Asia where forest clearings have led to large-scale fires in forested peatlands (van der werf et al., 2011). However, the physical link between the remotely sensed datasets and the observed phenological processes remains uncertain. For example, contrasting interpretations of MODIS Enhanced Vegetation Index (EVI: Huete et al., 2002) have been documented in relation to greening and browning during Amazonia drought periods (Atkinson, Dash, & Jeganathan, 2011; Huete et al., 2006; Samanta, Ganguly, & Myneni, 2011). Huete et al. (2006) confirmed a past eddy-flux study by Saleska et al. (2003) and observed an EVI increase of 25% with sunlight during dry seasons, a result that went against prevailing opinion that water stress during dry seasons and droughts should lead to a decrease in canopy photosynthesis. Huete et al. (2006) reasoned that sunlight, rather than the hydrological regime, was thus a more significant driver of the forest phenology and productivity. In contrast, Anderson et al. (2010) attributed these observed EVI increases to be caused by a decrease in shadows that result from pronounced tree mortality; the combined effects of dead stems and leaf-off trees leads to a decreased shade fraction contribution that in turn increases the EVI signal.

Remote sensing of tropical forests initially stemmed from the need to assess deforestation practices in developing countries for carbon emissions accounting, particularly from clear-cutting and fire (Achard et al., 2002; Cochrane, 2003). Many studies have utilized Landsat and SPOT high-resolution data in concert with aerial photographs to identify logging activities by their distinctive geometry (Asner et al., 2005; Skole & Tucker, 1993). Forest degradation has been characterized empirically from Advanced Very High Resolution Radiometer (AVHRR)-derived metrics such as NDVI, leaf-area index, land cover, land- surface temperature, albedo, and productivity (Achard et al., 2002; Lambin, 1999; Tucker, Holben, & Goff, 1984) that are not always tied to physical vegetation properties (Lambin, 1999). The widely researched interactions between climate, fire, and logging have also led to several studies using remote sensing to characterize patterns of area burned and forest land use in the Amazon. Cochrane et al. (1999) applied linear spectral unmixing to multi-temporal Landsat Thematic Mapper data and studies have also utilized Landsat-derived land cover classifications in assessments that specifically quantify vegetation vulnerability based on biophysical simulations (Hutyra et al., 2005 and Zhang, Justice, Jiang, Brunner, & Wilkie, 2006). Although there is clear utility of remotely sensed data to characterize burned areas, sources of confusion can arise in discriminating between intentional clearing (logging + fire) for agriculture or encroachment of wildfires from settlements and roads into intact forest (Cochrane et al., 1999).

In an initial *a posteriori* study, Nepstad et al. (1999) compared ground estimates of burned and logged areas to Landsat imagery and showed that ground assessments underestimated the affected area by half. This study then predicted the future vulnerability of these forests to fire by modeling soil moisture and leaf flammability under severe drought conditions; modeling demonstrated that in some scenarios these forests could be completely lost (Nepstad et al., 1999). In a later study, Nepstad et al. (2004) in *a posteriori* manner assessed the vulnerability of tropical forests to drought and fires from 1996–2001 by incorporating radiation inputs from the GOES-8 (Geostationary Operational Environmental Satellite) into the Penman-Montieth equation to estimate evapotranspiration impacts on soil moisture; ultimately an empirical relationship between LAI and soil moisture was applied as a means to predict tropical forest fire flammability. Given the widespread availability of LAI estimates, fractional photosynthetically active radiation (f_{PAR}), and light-use efficiency from contemporary satellite sensor data (e.g., the MODIS 15 Leaf Area Index (LAI) and f_{PAR} products, Hilker et al., 2011), one would expect that these and similar approaches would be readily applicable for *a priori* vulnerability assessments in tropical forests. Turner, Lambin, and Reenberg (2007) notes however that challenges will remain in quantifying resilience and vulnerability in tropical forests, including the maintenance of time series and the integration of multiple sensor platforms. Improved cloud masking has potential to alleviate problems with persistently partly cloudy scenes (Huang et al., 2010b). As with research in temperate ecosystems, studies that assess vulnerability in an *a priori* manner in tropical ecosystems have predominantly focused on evaluating the condition and function of ecosystems under future climatic scenarios (Diffenbaugh & Field, 2013; Mahli et al., 2009).

5. Remote Sensing of Vulnerability in Savannas and Semi-arid lands

Several studies have applied NDVI as a basis to assess vulnerability within savannas and semi-arid to arid lands (Easdale & Aguiar, 2012; Propastin, Forso, & Kappas, 2010). Considerable relevant research was conducted under the NASA SAFARI 92 and SAFARI 2000 field campaigns. The NASA Terrestrial Ecology Tree-Grass Project was a scoping study intended to advance the utility of remote sensing for measuring and monitoring mixed tree-grass ecosystems that are in large part defined by their high spatial and temporal heterogeneity. Remote sensing datasets incorporated into these studies have primarily been sourced from the AVHRR (Easdale & Aguiar, 2012; Propastin et al., 2010) and the Moderate-resolution Imaging Spectrodradiometer (MODIS) (Schaffrath, Barthold, & Bernhofer, 2011). NDVI is often applied as a surrogate for productivity, which in these systems may translate into the ecosystem service of forage for ungulates (Easdale & Aguiar, 2012). These ecosystem goods can be defined as vulnerable when the productivity of forage, crops, or timber falls below a level needed to breakeven financially or sustain a herd (Leurs, Lobell, Sklar, Addams, & Matson, 2003; Lobell, Ortiz-Monasterio, Addams, & Asner, 2002). Variations in NDVI as a proxy for productivity and meteorological observations have also been related to climatic variations (Anyamba & Eastman, 1996; Anyamba, Tucker, & Eastman, 2001; Christensen, Coughenour, Ellis, & Chen, 2004; Propastin et al., 2010).

In *a posteriori* study, Propastin et al. (2010) assessed the frequency of concurrent NDVI and El-Niño Southern Oscillation (ENSO) anomalies to evaluate the long-term sensitivity of vegetation areas and to produce a map of vegetation vulnerability for Africa. The study was hindered by insufficient land cover data and aggregating 8 km spatial resolution pixels to 24 km pixels (Propastin et al., 2010), arguably too coarse a spatial resolution for effective land management application. Similarly, the photochemical reflectance index (PRI; Gamon, Serrano, & Surfus, 1997) has been applied as an indicator of plant water and heat stress (Dobrowski, Pishnik, Zacro-Tejada, & Ustin, 2005; Saurez et al., 2008). The PRI is highly correlated with CO_2 uptake and photosynthetic radiation use efficiency at the leaf, canopy, and ecosystem levels within different plant functional types (Garbulsky, Peñuelas, Gamon, Inoue, & Filella, 2011; Hilker et al., 2011). As outlined in the next section, PRI is already being applied *a priori* to evaluate wetland vulnerability and therefore this and similar index-based approaches should be investigated further for prospective assessments of ecosystem service vulnerability.

Semi-arid steppe ecosystems have been widely studied with remotely sensed data (primarily using NDVI) to monitor changing ecosystem composition; however few studies have taken the step to specifically define potential change in terms of vulnerability metrics. For example, there has been considerable progress in monitoring and projecting the invasion of the annual grass *Bromus tectorum* and displacement of the native shrubland steppe utilizing Landsat time series (Bradley & Mustard, 2005; Peterson, 2005), including assessments of risk of invasion due to anthropogenic climate change (Bradley, 2009; Bradley & Mustard, 2006). The impacts of climate change on semi-arid and agricultural systems have the potential to be more profound than on any other vegetation type, as the inability of certain species to be productive could have considerable ramifications on the ability of that social-ecological-system to sustain a necessary forage or food source. A seminal *a priori* vulnerability assessment focusing on European landscapes (forests, Mediterranean, and agriculture) and bioenergy in Europe was conducted by Schröter et al. (2005). This study applied climate scenarios, ecosystem and process-based forest growth models, and stakeholder interviews to predict changes in Europe-wide cropland area and carbon budgets by 2080, where the changes were driven in part by human land-use decisions. The authors predicted a loss in the Europe carbon sink by 2050 and a conversation of agricultural to forest lands (Schröter et al., 2005).

6. Remote sensing of vulnerability in wetlands and mangroves

Although a wide body of research has focused on assessing the vulnerability of coral reefs and coastal regions to tsunamis, tidal surges, and flooding (Adger, Hughes, Folke, Carpenter, & Rockstorm, 2005; Eckert, Jelinek, Zueg, & Kruasmann, 2012; Klein & Nicholls, 1999; Kumar & Kunte, 2012; Nicholls, 2004; Romer et al., 2012), remote sensing for the vulnerability assessment of wetlands and mangroves are limited (Johnson et al., 2005; Li, Wang, Liang, & Zhou, 2006; Omo-Irabor et al., 2011; Simas et al., 2001). Detailed reviews of remote sensing methods to characterize wetlands, coastal vegetation, salt marshes and water inundation areas exist in the literature (Smith, 1997; Henderson & Lewis, 2008, Silva et al., 2008 and Klemas, 2011). The lack of attention paid to vulnerability of wetlands is surprising, especially given the 1971 Ramsar Convention of Wetlands that led to numerous research initiatives assessing wetland vulnerability to anthropogenic climate change and the associated broader biophysical and social-ecological implications (Covich et al., 1997; Grey & Sadoff, 2007). The need for assessments focusing on wetlands and riparian areas is heightened given a likely result of increased temperatures is the reduction of spring and summer run-off from depleted up stream snow-packs (Diffenbaugh & Field, 2013; Laurance et al., 2011).

The remote sensing of wetlands has predominantly focused on characterizing the type of wetland, rather than the vegetation occupying it. Current methodologies characterize wetlands via coupling thermal, optical, infrared, and RADAR data (Bourgeau-Chavez, Riordan, Powell, Miller, & Nowels, 2009; Bourgeau-Chavez et al., 2005; Rosenqvist, Shimada, & Watanabe, 2007). In terms of RADAR, C-band is often used for identifying wetlands with low-lying vegetation, while longer wavelength L-band RADAR is used for wetlands with taller vegetation (Kasischke, Melack, & Dobson, 1997; Whitcomb, Moghaddam, McDonald, Kellndorfer, & Podest, 2009). In a study to evaluate vulnerability in the upper reaches of the Minjiang River of China, Li et al. (2006) combined Landsat MSS data with topography, soil, and climatic records.

Notable exceptions are studies utilizing remote sensing and spatial datasets to evaluate the vulnerability of coastal vegetation ecosystems to sea-level changes and disturbances including oil spills, tsunamis, and hurricane-induced storm surges (Cooper et al., 2013; de Andrade, Szlafsztein, Souza-Filho, dos Reis Araujo, & Gomes, 2010; Omo-Irabor et al., 2011; Villa, Boschetti, Morse, & Politte, 2012). Often, these assessments are a response to specific disaster events (e.g., the 2005 tsunami off the coast of Indonesia). The application of data from laser altimetry or other data sources coupled within a GIS to predict which areas will be flooded under future scenarios is an effective, *a priori* utility of remote sensing data to evaluate future vulnerabilities. Cooper et al. (2013) applied scenarios of sea-level rise to LiDAR based surface models of Hawaii to predict potential vulnerability. Other studies have applied both bathymetry and Landsat-based NDVI measures to map the extent of salt marsh vulnerability to sea-level rise, defining vulnerability as the percentage of salt marsh area likely to be impacted under future sea-level rise scenarios (Simas et al., 2001). PRI has been demonstrated to exhibit a strong response to salinity exposure and has considerable potential as a candidate *a priori* approach for characterizing plant stress in mangroves in response to both freshwater flooding and soil salinity (Naumann, Young, & Anderson, 2008; Nicholl, Rascher, Matsubara, & Osmond, 2006; Song, White, & Heimann, 2012; Zinnert, Nelson, & Hoffman, 2012).

For mangrove forests, vulnerability to climate change results from the inability of these ecosystems to respond to changing sea level elevations or higher intensity of ocean disturbances (Cooper et al., 2013; Gilman et al., 2008; Villa et al., 2012). Remote sensing datasets have a clear utility to map the elevation changes in sea levels and surrounding landscapes over time and determine the areal extent of mangroves under risk. Visual interpretation of high spatial resolution observation, such as from IKONOS imagery, assisted in the development of a vulnerability map of potential oil spill impacts on mangroves and other coastal landscapes (de Andrade et al., 2010). Landsat series and Shuttle Radar Topography Mission data have also been used to map the spatial extent of mangroves and other vegetation types (Omo-Irabor et al., 2011); although such datasets are often used in combination with social data layers and climate to assess overall vulnerability of the ecosystem (de Andrade et al., 2010; Omo-Irabor et al., 2011).

7. Remote sensing of vulnerability in boreal forests and tundra

There is considerable interest in understanding northern latitude ecosystems, as emphasized by large initiatives like the Boreal Ecosystem-Atmosphere Study (BOREAS, Sellers et al., 1997; Hall, 1999; Hall, 2001), the Land-Air-Ice Interactions (LAII) program (Stow et al., 2004), the International Polar Year-Back to the Future (IPY-BTF) project (Callaghan, Tweedie, Akerman, et al., 2011; Callaghan, Tweedie, & Webber, 2011), and the soon to be initiated Arctic Boreal Vulnerability Experiment (ABoVE, NASA Terrestrial Ecology Program). This interest stems from numerous observations that northern latitude ecosystems are already experiencing the impacts of anthropogenic climate change, and will continue to be impacted more rapidly and at greater magnitudes than lower latitude ecosystems (Ford, Smith, & Wandel, 2006; Kasischke et al., 2010; Kittel et al., 2011; McGuire et al., 2006, 2009).

Remotely sensed data have been widely used to characterize vegetation condition and disturbances in northern latitude ecosystems (Beck et al., 2011; Goetz, Fiske, & Bunn, 2006; Goetz & Prince, 1996; Jones et al., 2009; Kasischke et al., 2010; Kolden & Rogan, 2013; Stow et al., 2004). The primary ecosystem service that has been addressed is carbon storage, as boreal ecosystems are estimated to contain nearly one-third of the carbon sequestered in global terrestrial ecosystems (Apps et al., 1993). In boreal forests and peatlands, vulnerability is usually represented as the potential for an ecosystem shift; particularly from cooler, wetter spruce-dominated forests that store large volumes of terrestrial soil carbon to warmer, drier mixed forests and shrublands that sequester less carbon and are more susceptible to carbon-releasing wildfires (Kasischke et al., 2010). The ability to quantify vegetation productivity, ecosystem vulnerability to disturbance, and the spatially and temporally heterogeneous impacts of fire with remotely sensed data is critical to characterizing the impacts of anthropogenic climate change on carbon sequestration in boreal ecosystems (Balshi et al., 2009; Beck et al., 2011; Kasischke et al., 2010; Kolden & Abatzoglou, 2012; Lentile et al., 2006).

The majority of studies have used NDVI time series from AVHRR, and more recently MODIS, to measure boreal forest change and dynamics (Beck & Goetz, 2011; Beck et al., 2007; Berner, Beck, Bunn, Lloyd, & Goetz, 2011; Goetz et al., 2005; Piao et al., 2011; Rigina, 2003). In tundra ecosystems, similar approaches have been used to assess vegetation change where an important component of vulnerability is the magnitude and rate of conversion to shrublands (Beck & Goetz, 2011; Beck et al., 2007; Epstein, Raynolds, Walker, Bhatt, & Pinzon, 2012; Lin, Johnson, Andresen, & Tweedie, 2012; Stow et al., 2004). Associated with each of these aspects are the rates of change of greenness and brownness (Olthof, Pouliot, Latifovic, & Chen, 2008; Raynolds, Walker, Epstein, Pinzon, & Tucker, 2012; Raynolds, Walker, & Maier, 2006; Stow et al., 2004; Verbyla, 2008; Walker et al., 2012; Zhang et al., 2008; Zhou et al., 2001) that drive gross changes in vegetation productivity, composition, and function across taiga and tundra systems, for example, movement of the tundra-taiga boundary (Callaghan, Werkman, & Crawford, 2002; Ranson, Sun, Kharuk, & Kovacs, 2004; Rees, Brown, Mikkola, Virtanen, & Werkman, 2002). A critical measure of vulnerability in all of these systems are phenological changes (Bunn & Goetz, 2006; Zhang et al., 2008) that affect the seasonality of physical, ecological, and social processes in northern latitudes.

In *a posteriori* study, Forbes et al. (2009) evaluated the spatial resolution necessary to visibly detect potential vulnerabilities of a nomadic subsistence community in northern Siberia and quantified the total area of impacts from those indicators using ASTER, Landsat TM and MSS, and panchromatic Quickbird 2 imagery. Others have utilized remotely sensed products (e.g., NDVI, NPP, land cover, land surface temperature, sea surface temperature) derived from MODIS or synthetic aperture radar (SAR), and high spatial-resolution aerial photographs to assess *a posteriori* vulnerabilities of vegetation to anthropogenic climate change (Alessa, Kliskey, & Brown, 2008; Alessa, Kliskey, Lammers, et al., 2008; Alessa, Kliskey, White, Busey, & Hinzman, 2008; John et al., 2004; Kofinas et al., 2010; Ray, Kolden, & Chapin, 2012; Sturm et al., 2001, 2005). MODIS NDVI and NPP products have also been applied in other studies seeking to characterize species diversity impacts in Alaska, changing habitat conditions for herbivores, and interactive effects of changing hydrology on vegetation (Alessa, Kliskey, & Brown, 2008; Alessa, Kliskey, White, Busey, et al., 2008; Stow et al., 2004). The considerable research in these ecosystems has led to studies combining social science data with remotely sensed imagery such as Landsat, MODIS and Quickbird to create various integrated vulnerability assessment tools (e.g., The Arctic Water Resources Vulnerability Index, AWRVI, Alessa, Kliskey, Lammers, et al., 2008) that can be used as an *a priori* management tool.

8. Development of early warning systems

Table 2 presents example and proposed remote sensing metrics to conduct *a priori* assessments of vulnerability. With the definition of such metrics, it follows that it would be feasible to use remotely sensed spatial layers to construct an ecologically-based modeling and decision support framework that enables more accurate forecasting of when and where future vulnerabilities are likely to occur. Such systems are often described as an early warning system (EWS: Basher, 2006; Huggel et al., 2012; Fig. 3). As simulations are modeled, the EWS can identify causes, and when in time and space a tipping point will be reached. In Fig. 3, the left pointing truncated arrow indicates the decision to restart the simulation process under a different set of initial conditions. Such adaptation scenarios enable the evaluation of how to mitigate or even prevent the system reaching the undesirable condition. Even in the most optimistic scenario it is unlikely that all future undesirable impacts can be prevented; rather the goal will be conduct mitigation and adaptation strategies to either minimize the loss of the desired ecosystem service or identify alternative regimes/states where other equally viable ecosystem goods and services can be obtained (Fig. 1). When properly constructed, an EWS can help land managers identify geographic and temporal locations for targeted management actions. For instance, fire early warning systems have been implemented in several countries (de Groot et al., 2006), and recent efforts in western North America have focused on identifying forest vulnerability to mountain pine beetle infestations such that management actions can mitigate potential negative impacts (Wulder, White, Carroll, & Coops, 2009).

The application of metrics to create such imagery-driven ecological early warning systems is not a new concept (Cairns & Vanderschalie, 1980; Scheffer et al., 2009); remote sensing and GIS have been combined in this format to inform complex management decisions for decades. For instance, the Famine EWS (FEWS) was developed in the 1980s by USAID to provide early detection of drought and subsequent

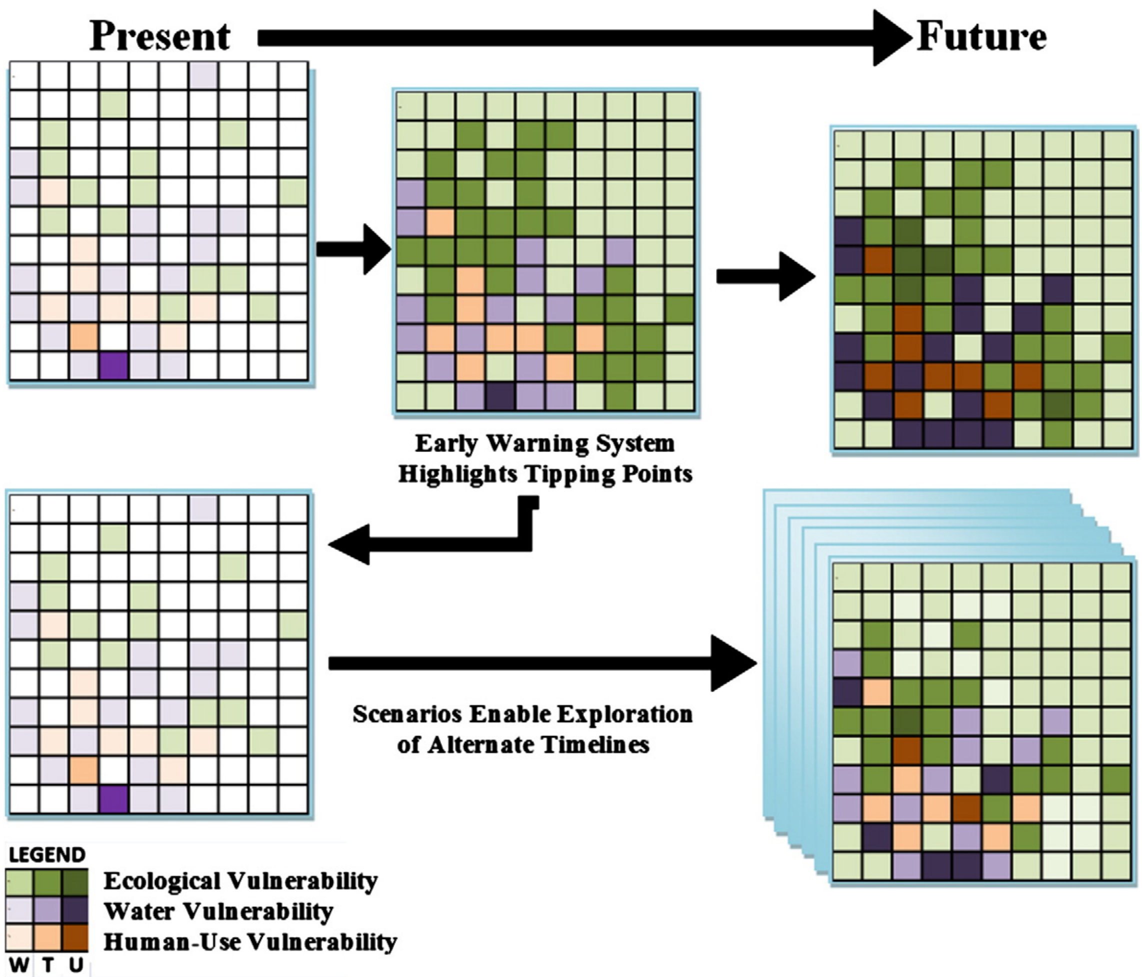

Fig. 3. The utility of an early warning system (EWS) as highlighted via a vulnerability, impacts, and adaptation (VIA) framework. In this example, W = Warning, T = Tipping Point/Critical Threshold, and U = Unrecoverable (Table 3).

crop failure and food insecurity in Africa using AVHRR data (Lozano-Garcia et al., 1995; Brown, 2008) and continues 25 years later in other at-risk regions globally using MODIS. The Northwest Forest Plan developed by the United States Forest Service in the mid-1990s has been used to quantify critical forest habitat and address the impacts of forest management and natural disturbances (Moeur et al., 2005). In the mid-Atlantic region of the United States, remotely sensed (Landsat) estimates of land use and land cover play an important role in management decisions in the 30-year effort to improve water quality within the Chesapeake Bay Watershed (Jantz, Goetz, & Jantz, 2005). A similar effort was used to understand how alternate land management scenarios would impact ecosystem services within the Willamette River Basin of western Oregon in the northwestern United States (Baker et al., 2004). Additionally, several landscape modelling scenario tools have been developed and include LANDIS-II, Forest Vegetation Simulator (FVS), and EVOLAND/ENVISION (Bolte, Hulse, Gregory, & Smith, 2007; He & Mladenoff, 1999; Karam, Weisberg, Scheller, Johnson, & Miller, 2013). Fire danger rating is calculated integrating vegetation status information, and basic meteorological data, to provide early warning of the potential for severe wildfires (de Groot et al., 2006). The fire weather index (FWI), initially developed as part of the Canadian Forest Fire Danger Rating System (Van Wagner, 1987), has been modified and calibrated to be used in fire early warning systems in Europe (López, San-Miguel-Ayanz, & Burgan, 2002), boreal forests and tropical forests (Taylor & Alexander, 2006).

More recently, the application of remote sensing and GIS has been highlighted as a means to significantly advance the science of ecological EWS (Niemi & McDonald, 2004). In contemporary studies, the application of these techniques is often considered the last stage of vulnerability, impacts, and adaptation assessments (Huggel et al., 2012). However, this is not necessarily the most optimal approach. Through understanding how systems become vulnerable (or, conversely, are resilient) it becomes feasible to model alternate possible future scenarios through a better understanding of feedbacks between biogeophysical and socio-economic variables. Remote sensing data have been applied within EWS to detect areas susceptible to tsunamis (Wang & Li, 2008) and landslides (Hong & Adler, 2007; Huggel et al., 2012) and identify forests that are susceptible to changes in phenology due to disturbances, insects, and diseases (Daterman, Wenz, & Sheehan, 2004; Hargrove, Spruce, Gasser, & Hoffman, 2009), among other stress agents. Specifically, Hargrove et al. (2009) applied a time series of NDVI to evaluate rapid changes in forest phenology associated with a disturbance or insect/disease outbreak. In the realm of ecosystem goods and services, the EWS approach is relatively underdeveloped (Alessa et al., 2008).

We argue that an EWS will have value only if it has the following key characteristics:

1. The ability to provide warnings at spatial and temporal scales that are relevant to the tools of land management (Nitschke & Innes, 2008).
2. The ability to alert managers before the ecosystem reaches a state of decay from which no reasonable action could be implemented to change the predicted outcome (i.e. "Warning" in Table 3, Fig. 3).
3. The metrics should be general and transferable to similar ecosystems experiencing similar concerns. Preserving the biophysical link between variables that can be quantified remotely (i.e. reflectance, cover, emittance, elevations, etc.) and an observable impact on the ground will allow transferability even when sociocultural and economic conditions vary.
4. To fully capture the dynamics and complexity of social-ecological systems, Early Warning Systems will need to include data relating to social values, cultures, and economies among other variables (Alessa, Kliskey, & Brown, 2008; Alessa, Kliskey, Lammers, et al., 2008; Alessa, Kliskey, White, Busey, et al., 2008; Basher, 2006).
5. The rationale behind the methods and metrics must be both rigorous enough to satisfy scientists' standards for accuracy, objectivity, repeatability, and transparency, yet accessible enough to convince land managers and stakeholders in management actions.
6. For some purposes, the spatial data must be placed in the context of various social components that are associated with it. For example, understanding why vegetation clearing has occurred, say for purposes of development, allows insight as to whether such change is on-going or isolated, expected or not, and whether it can be managed with existing technical and policy tools.
7. Method transparency is essential as studies seeking to evaluate vulnerability should present sufficient methodological detail to enable replicated studies, especially if specific sensors become outdated or unavailable.

For the assessment of ecosystem goods and services, the importance of item (1) cannot be overstated as EWS must account for feedbacks associated with social processes (e.g., land-use, zoning) that are part of land-use planning. In order to be useful, the EWS must comprehensively synthesize the breadth and depth of ecosystem goods and services that are of interest to managers and broader user groups such as local communities. We propose that such considerations are vastly underestimated and often not factored into EWS or ecosystem assessments, rendering them of limited use for long term decision and policy making.

Ecosystem goods and services are somewhat artificial terms, reflecting primarily an economic/market approach that assumes a value can be placed on all components of an ecosystem (Seppelt, Dormann, Eppink, Lautenbach, & Scmidt, 2011). This may limit the utility of EWS if this is the only perspective because we know that a) humans are not rational, b) culture can trump economics and c) the values held in a given community drive the way decisions are made more readily than do objective facts (Alessa, Kliskey, & Brown, 2008). EWS must incorporate the potential response of a set of diverse stakeholders to specific biotic and abiotic scenarios. For example, many invasive species are highly valued by residents (e.g., the May Day Tree or chokecherry, genus *Prunus*, in parts of Alaska), which renders agency arguments and policy toward elimination impotent. Ultimately, remote sensing data give the investigator insight as to where to concentrate inquiry regarding changing social dynamics relative to changing ecosystems. Such a tool will increase the precision and utility of the emerging field of social ecological systems science.

The development of EWS and overall use of remote sensing techniques to quantify ecosystem vulnerability could also be strengthened by the increased use of temporal datasets. Thus far, the majority of vulnerability studies that have used remote sensing data have not explicitly

Table 3
Typical Vulnerability Matrix Framework (adapted from Alessa, Kliskey, & Brown, 2008). The threshold breaks, M_i, are typically selected using distributional breaks (e.g., quintiles). Equally, however physical breaks could be selected based on known threshold values (e.g., ecophysiological thresholds below which growth does not occur).

Vulnerability Rating			
Healthy	1.0	$>M_4$	A highly resilient system with a surplus of all desired ecosystem goods services
Sustained	0.75	M_3–M_4	A moderately resilient system with all desired ecosystem goods and services without net loss
Warning	0.50	M_2–M_3	Ecosystem goods and services produced but with a net loss of function; management actions needed
Critical	0.25	M_1–M_2	The tipping point: A vulnerable system with partial ecosystem goods and services produced but with a net loss, where radical management actions would be needed
Unrecoverable	0.00	$<M_1$	A highly vulnerable system which has degraded to a point where ecosystem goods and services are lost, nor can any reasonable management actions be performed to restore the ecosystem goods and services

included temporal datasets. However, remote sensing data records are readily available at greater than decadal time-scales (i.e. AVHRR and Landsat: 30 + years, aerial photography: 100 + years) and the inclusion of temporal series data to enhance vulnerability studies has the potential lead to a significantly different finding or add an extra dynamic dimension that single date analysis cannot capture (Gutman & Masek, 2012). Incorporating temporal series data allows studies to evaluate inter-annual variability as well as magnitudes and rates of change associated with coupled bio-climatic and social processes. By adding this temporal dimension, remote sensing datasets have clear value in providing consistent and quantitative indicators relating to the cover, structure, condition, temporal trends, and spatial patterns of vegetation. These properties are often linked to the system's exposure and adaptive capacity, although social-economic and political measures are less accessible (Taubenbock et al., 2008).

9. Conclusions

To recap, the questions we sought to address were:

(1) Can remote sensing datasets be used to evaluate vulnerability of ecosystems traits such as species composition, structure, and function?
(2) Can remote sensing datasets be used to effectively evaluate vulnerability of critical ecosystem goods and services?
(3) Can we detect vulnerability of ecosystem goods and services before a disturbance occurs? Or conversely, can remote sensing only be used to evaluate antecedent conditions and recovery trajectories after a disturbance has occurred?

In terms of question (1), past reviews highlighted the potential and limitations of remote sensing to characterize species composition, structure, and function (Ustin & Gamon, 2010). Although we identified several studies that applied metrics of ecosystem traits within vulnerability assessments, a larger number of studies used metrics that did not exhibit strong mechanistic connections between the impacts of climate change and the vegetation within the ecosystems. A potential *a priori* route for further research that connects remote sensing of ecosystem traits with future ecosystem conditions is to use remote sensing data to characterize current successional stage, or where on a likely successional temporal trajectory, an ecosystem is. Remote sensing studies have characterized succession stage in a variety of ecosystems using diverse datasets such as LiDAR and spectral imagery (Falkowski, Evans, Martinuzzi, Gessler, & Hudak, 2009; Hall, Botkin, Strebel, Woods, & Goetz, 1991; Petit, Scudder, & Lambin, 2001), usually within classification and regression tree type methodologies (Bergen & Dronova, 2007; Falkowski et al., 2009). Identification of where in a successional pathway a specific ecosystem is may provide valuable information of the degree to which an ecosystem is more susceptible to the occurrence of heightened impacts of disturbances; i.e. whether the ecosystem is within a stable or vulnerable condition (Fig. 1). Another potential ecosystem trait to explore is the range of vegetation canopy cover under which a desired ecosystem service is sustainable. In tropical forests, reduced canopy cover is often considered a metric of vulnerability highlighting an increased probability of damage from future fires, while in temperate forests early warning systems already exist that indicate tipping points in the production of merchantable timber based on the proportion of insect infected trees (Hargrove et al., 2009), and in semi-arid rangelands increases in woody plant encroachment will lead to an upper canopy cover limit above which the non-tree species (e.g., grasses used for cattle forage) are unsustainable (Archer, 1990; Hudak, 1999).

In terms of question (2), remotely sensed metrics exist that can be readily tied to common vegetation ecosystem goods and services (Tables 1 and 2). A notable and generally transferrable approach was observed in agroforestry, where studies projected under what future conditions certain crops are not be able to sustain specific levels of ecosystem goods and services (Leurs et al., 2003; Lobell et al., 2002; Schröter et al., 2005). Similarly, recent studies by Smith, Cleveland, Reed, Miller, and Running (2012), Smith, Zhao, and Running (2012) and Smith, Zhao, and Running (2012) applied remote sensing assessments of net primary productivity to the United States, and globally, to highlight the maximum amount of bioenergy that could be sustainably acquired from terrestrial vegetation. The assessment of such sustainable maximums from crops and bioenergy could be extended to other ecosystem goods like forest timber and rangeland forage. Such assessments represent research that can be readily applied to conduct *a priori* assessments via remote sensing data. This approach could equally be widely applied to ecosystem goods and services related to monitoring temporal means of water quality, timing, and yield and may enable a general framework to assess vulnerability at the interface of science disciplines.

In terms of question (3), several studies applied remote sensing in a prospective mode to predict vulnerabilities. Notably, studies coupled plant physiology metrics with future modeling scenarios to identify vulnerable ecosystems based on which species will likely grow and their health or stress (Coops & Waring, 2011; Coops et al., 2012; Waring et al., 2011; Zinnert et al., 2012). Similar approaches could evaluate the probability of species re-establishing following a disturbance. Given the widespread availability of remote sensing studies that have focused on evaluating early signs of vegetation stress from water, nutrients, and heat (Eitel, Gessler, Smith, & Robberecht, 2006; Eitel, Long, Gessler, & Smith, 2007; Hilker et al., 2011), it is apparent that such approaches would be readily applicable for *a priori* vulnerability assessments across global ecosystems. Equally, predictive modeling of "what-if" scenarios is widely applicable beyond remote sensing applications. The application of laser altimetry coupled within a GIS to predict which coastal areas may be flooded under future scenarios was a simple, but effective, *a priori* utility of remote sensing data to evaluate future vulnerabilities of vegetation, land planning, and hydrology (Cooper et al., 2013). Similarly, the application of laser altimetry and other active remote sensing data can be readily coupled with hydrological, microclimatic, and physiological models to predict snow-rain transition zones and subsequent negative impacts on summer water availability (Cayan, Kammerdiener, Dettinger, Caprio, & Peterson, 2001; Mote, Hamlet, Clark, & Lettenmaier, 2005; Tinkham et al., 2013).

Although the potential of remote sensing to provide *a priori* information of future vulnerabilities has been demonstrated through specific examples, we contend that the wider remote sensing community needs to further investigate potential prospective approaches. The application of metrics that have clear mechanistic links between both remotely sensed metrics of vegetation that are sensitive to the changes in climate and the desired ecosystem goods and services will facilitate the development of a robust EWS. Ideally, methodological frameworks are needed that can be applied to a wide array of disciplines with the ultimate goal to provide additional information layers to land management personnel that are charged sustaining ecological goods and services. It is unlikely that one could ever replace vulnerability assessments that are largely based on social surveys, with maps purely derived from remote sensing. However, given that many vulnerability assessment systems already integrate multiple data sources via a weighting schema, it seems reasonable to propose that geospatial data of ecosystem vulnerability that are produced with lower uncertainties could be given higher weights than aspatial data produced with higher uncertainties.

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